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Full-Scale Field Evaluation of Waste  
Disposal from Coal-Fired Electric  
Generating Plants. Volume 1  
Sections 1 through 5

Little (Arthur D.), Inc., Cambridge, MA

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FULL-SCALE FIELD EVALUATION OF  
WASTE DISPOSAL FROM COAL-FIRED  
ELECTRIC GENERATING PLANTS  
Volume I. Sections 1 Through 5

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# **TECHNICAL REPORT DATA**

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| 16. ABSTRACT<br>The six-volume report summarizes results of a 3-year study of current coal ash and flue gas desulfurization (FGD) waste disposal practices at coal-fired electric generating plants. The study involved characterization of wastes, environmental data gathering, evaluation of environmental effects, and engineering/cost evaluations of disposal practices at six sites around the country. Study results provide technical background data and information for EPA, state and local permitting officials, and the utility industry for implementing environmentally sound disposal practices. Study data suggest that no environmental effects have occurred at any of the six sites; i. e., data from wells downgradient of the disposal sites indicate that waste leachate has resulted in concentrations of chemicals less than the EPA primary drinking water standards. A generic environmental evaluation--based on a matrix of four waste types, three disposal methods, and five environmental settings--shows that, on balance, technology exists for environmentally sound disposal of coal ash and FGD wastes for ponding, interim ponding/landfilling, and landfilling. For some combinations of waste types, disposal methods, and environmental settings, mitigation methods must be taken to avoid adverse environmental effects. Costs of waste disposal operations are highly system and site specific. |  |                                                                      |  |                                                         |  |
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| Pollution                      Ashes                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                         |  | Pollution Control                                                    |  | 13B                      21B                            |  |
| Waste Disposal              Flue Gases                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                       |  | Stationary Sources                                                   |  | 14G                      07A, 07D                       |  |
| Wastes                          Desulfurization                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                              |  |                                                                      |  | 21D                      14A                            |  |
| Coal                              Cost Engineering                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                           |  |                                                                      |  | 10B                                                     |  |
| Combustion                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                   |  |                                                                      |  |                                                         |  |
| Electric Power Plants                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                                        |  |                                                                      |  |                                                         |  |
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## ABSTRACT

This report summarizes results of a 3-year study of current coal ash and flue gas desulfurization (FGD) waste disposal practices at coal-fired electric generating plants. The study was conducted by Arthur D. Little, Inc., under EPA contract 68-02-3167, and involved characterizing wastes, gathering environmental data, assessing environmental effects, and evaluating the engineering/costs of disposal practices at six selected sites in various locations around the country. Results of the study are providing technical background data and information to EPA, State and local permitting officials, and the utility industry for implementing environmentally sound disposal practices.

Data from the study suggest that no major environmental effects have occurred at any of the six sites. For example, data from wells downgradient of the disposal sites indicate that the contribution of waste leachate to the groundwater has generally resulted in concentrations of chemicals less than the primary drinking water standards established by EPA. Although occasional exceedances of the standards were observed, these were not necessarily attributable to coal ash and FGD waste. A generic environmental evaluation based on a matrix of four waste types, three disposal methods, and five environmental settings (based on climate and hydrogeology) shows that technology exists for environmentally sound disposal of coal ash and FGD wastes for ponding, interim ponding/landfilling, and landfilling. For some combinations of waste types, disposal methods, and environmental settings, measures must be taken to avoid adverse environmental effects. However, site-specific application of good engineering design and practices can mitigate most potentially adverse effects of coal ash and FGD waste disposal. Costs of waste disposal operations are highly system- and site-specific.

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## SECTION 6

### ENGINEERING COST ASSESSMENT OF COAL ASH AND FGD WASTE HANDLING AND DISPOSAL SYSTEMS

#### 6.1 ENGINEERING OF COAL-FIRED UTILITY SOLID WASTE HANDLING AND DISPOSAL SYSTEMS

Coal ash and FGD waste disposal operations consist of a number of activities which include the handling of wastes from the point of their collection up to and through their ultimate disposal. Variations in collection and disposal systems necessitate a variety of waste handling, processing, storage and transport alternatives. Additionally, variations among waste types call for different approaches to waste management.

The engineering and cost data contained in this section were assembled with the modular approach outlined in Section 4.7. In general, a matrix of waste types and waste management activities was developed. For each waste type/waste management activity combination one or more engineering design option is presented. It is the combination of the appropriate modular options which leads to a preliminary engineering specification that forms the basis for the conceptual cost estimate for the integrated waste management system.

##### 6.1.1 Coal-Fired Utility Solid Waste Handling and Disposal Options

Many options are available for handling and disposing of coal combustion solid wastes. However, all coal-fired utility waste handling and disposal operations can be divided into subsystems which fall into a relatively small number of broad categories. This categorization is the basis for the modular approach adopted for this engineering assessment. Five categories (denoted modules for the purposes of this study) of coal-fired utility solid waste handling and disposal activities have been identified:

- Raw Materials Handling and Storage;
- Coal-Fired Utility Solid Waste Handling and Processing;
- Coal-Fired Utility Solid Waste Storage;
- Coal-Fired Utility Solid Waste Transport; and
- Coal-Fired Utility Solid Waste Placement and Disposal (includes Site Monitoring and Reclamation).



Within each module there exist a number of system design options which vary according to the waste type (i.e., fly ash, bottom ash and FGD waste) and the existing or desired physical states of the waste (i.e., dry solid wastes, aqueous slurries of solid wastes and dewatered solid wastes). In this engineering assessment, an attempt has been made to elucidate differences which can be identified within modules. The five modules can thus be viewed as five collections of engineering design alternatives. Selection of alternatives from each module which satisfy the general disposal and handling needs of a particular site results in a preliminary process specification. In some cases, more than one alternative within a given module can be required for different waste types or for wastes collected in different physical states. An overview of the five modules and the alternatives within each of the modules are presented schematically in Figures 6.1 and 6.2 for coal ash and FGD waste, respectively. As shown, some modules do not apply to all waste types. Table 6.1 lists waste types and the modules which apply to each.

In developing engineering and cost data on utility solid waste handling and disposal systems, it is difficult to divorce oneself entirely from the other aspects of the environmental control system. Hence, broad process data were developed for auxiliary areas that tend to shed substantial light on the design and cost of environmental control, including:

- the air pollution control system, which may consist of particulate and, in some cases, sulfur oxides control; and
- auxiliary systems for handling wastes other than coal ash or FGD wastes which enter the solid waste handling and disposal system.

The rest of Section 6.1 provides an overview of engineering data on the modules and auxiliary areas discussed above.

#### 6.1.2 Coal-Fired Utility Solid Waste Collection

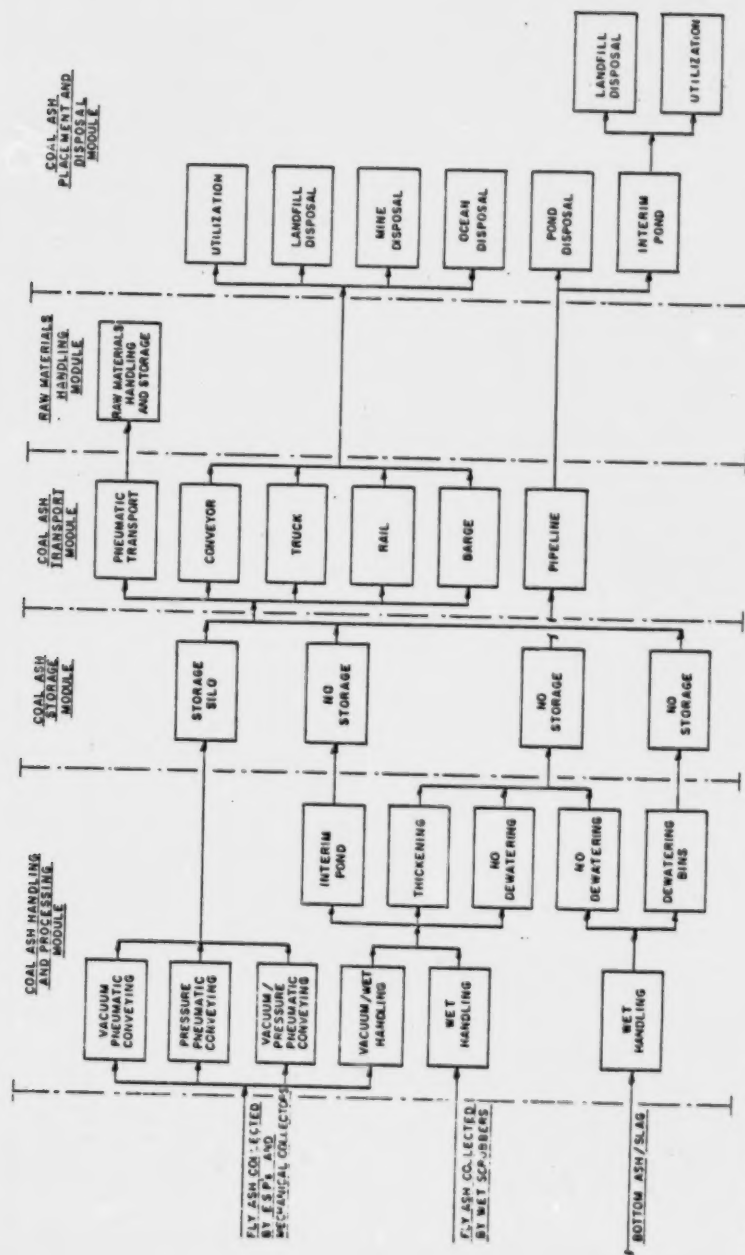
##### 6.1.2.1 Particulate Control Technology--

Control of particulate emissions from coal-fired boilers can be achieved by several types of particulate control devices:

- mechanical dust collectors;
- wet scrubbers;
- electrostatic precipitators (ESPs); and
- fabric filters.

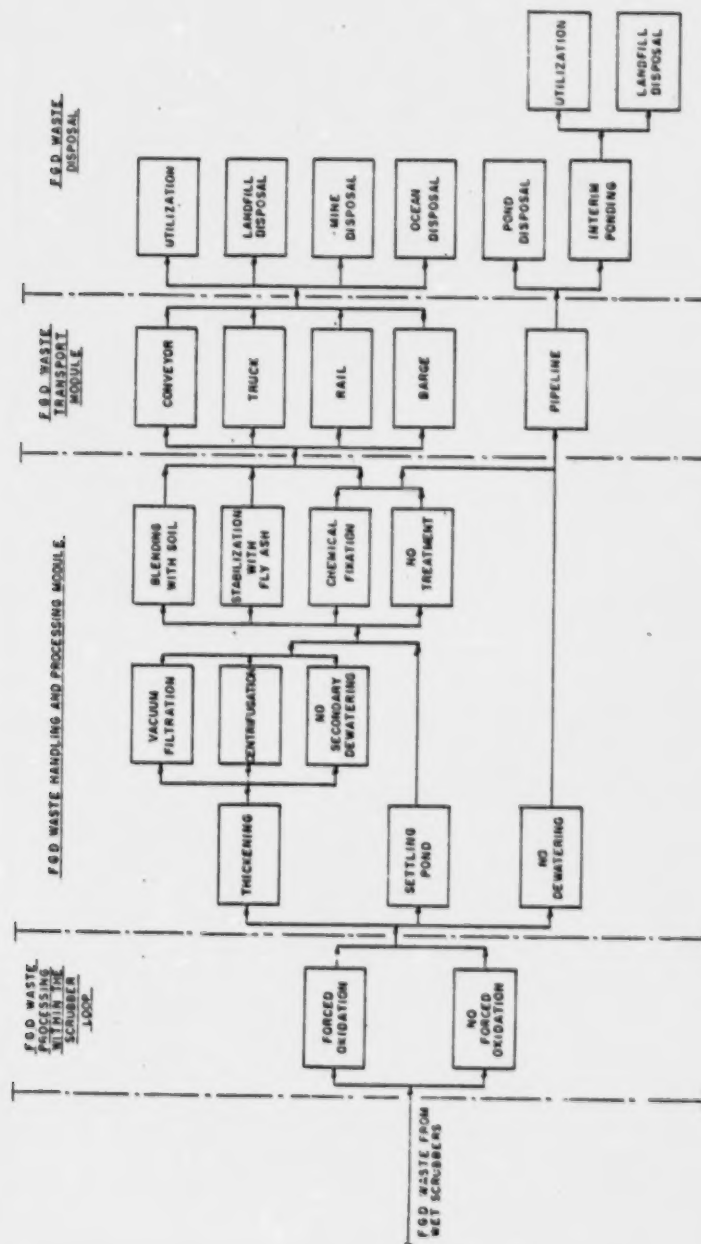
By far, ESPs are the most common particulate control devices employed by coal-fired electric generating utilities. Fabric filtration is more recently gaining acceptability for this application. The use of mechanical dust collectors, with fly ash collection efficiencies substantially lower than those of the other particulate control devices discussed here, is anticipated





Source: Arthur D. Little, Inc.

FIGURE 6.1 OVERVIEW OF WASTE HANDLING AND DISPOSAL OPTIONS FOR COAL ASH



Source: Arthur D. Little, Inc.

FIGURE 6.2 OVERVIEW OF WASTE HANDLING AND DISPOSAL OPTIONS FOR FGD WASTE

TABLE 6.1  
MATRIX OF WASTE TYPES AND WASTE HANDLING/DISPOSAL MODULES

| MODULE                                                 | WASTE TYPE |            |           |
|--------------------------------------------------------|------------|------------|-----------|
|                                                        | Fly Ash    | Bottom Ash | FGD Waste |
| Raw Materials Handling and Storage                     |            |            | ✓         |
| Coal-Fired Utility Solid Waste Handling and Processing | ✓          | ✓          | ✓         |
| Coal-Fired Utility Solid Waste Storage                 | ✓          |            |           |
| Coal-Fired Utility Solid Waste Transport               | ✓          | ✓          | ✓         |
| Coal-Fired Utility Solid Waste Placement and Disposal  | ✓          | ✓          | ✓         |

Source: Arthur D. Little, Inc.

to decline in the future due to implementation of more stringent air quality standards. Scrubbers are not often used for particulate control at utility boiler installations unless they are used for simultaneous particulate and sulfur dioxide removal or, when fly ash is used, as a source of alkalinity in FGD systems (e.g., fly ash alkali scrubbing) (12).

Electrostatic Precipitators (ESPs) -- Collection of particulates (fly ash) by electrostatic precipitation entails the following sequence of operations:

1. Fly ash particles become charged by colliding with gaseous ions and electrons generated in a high intensity corona discharge.
2. In the presence of an applied electric field, charged fly ash particles migrate toward a grounded collection electrode.
3. Collected fly ash material is removed from the plate by periodic rapping.

Modern ESPs are capable of achieving relatively high collection efficiencies (in some cases in excess of 99 percent). However, a number of parameters affect collection efficiency, including:

- precipitator size (i.e., specific collection area);
- applied collection field;
- gas temperature;
- electrical resistivity of the particulate matter;
- particle size; and
- gas composition.

The first three items are variables which may be adjusted in the design of an ESP. Electrical resistivity, particle size distribution and gas composition, however, depend on boiler operating conditions and the characteristics of the coal fired to the boiler. Gas composition can be altered by the introduction of additives which can subsequently reduce electrical resistivity of the particulates and thereby increase collection efficiency.

In the design of an ESP, increases in specified collection efficiency result in exponential increases in precipitator size. Thus, there is a trade-off between size (i.e., cost) and desired emission levels. Similarly, collection field requirements also increase with increasing efficiency specifications. Higher gas temperatures decrease the electrical resistivity of fly ash and, hence, increase collection efficiency. However, higher temperatures affect the volume of gas to be handled. Thus, an optimum efficiency can be obtained by selecting the proper operating temperature.

The electrical resistivity of fly ash has a significant influence on collection efficiency by electrostatic precipitation. High resistivity will decrease collection efficiency significantly. However, the electrical resistance of dust particles can be altered somewhat by altering the surrounding environment. As discussed previously, increases in temperature decrease the resistivity of fly ash particles. Similarly, the sulfur content of flue gas affects fly ash resistivity, since a layer of condensed, electrically conductive sulfuric acid can form on the surface of fly ash and reduce electrical resistance. Since the sulfur originally present in a given coal has a positive effect on fly ash collection efficiency, fly ash generated as a result of burning low sulfur coal will be more difficult to collect by electrostatic precipitation than that originating in a boiler burning high sulfur coal. For coals and fly ash having relatively high sodium contents, resistivity is controlled primarily by sodium concentration. In general, fly ash resistivity decreases with increasing coal alkali content.

Two types of ESPs are presently used for coal-fired utility applications, the cold-side ESP and the hot-side ESP. The basic design of both devices is similar except that hot-side ESPs are located on the hotter, 205 to 315°C (400 to 600°F) upstream side of the air preheater, while cold-side ESPs are located downstream, where the temperature is 120 to 205°C (250 to 400°F). This difference renders these devices suitable for different services; cold-side ESPs are primarily used to collect fly ash of low electrical resistance, while hot-side ESPs collect high resistivity fly ash by decreasing its resistivity due to the comparatively higher temperature of the collection environment. Hot-side ESPs, despite their ability to handle high resistivity particulates, have some serious disadvantages:

- Gas flows (volumes) are 50 percent higher than in cold-side systems because of the expansion of gases at higher temperatures.
- Precipitation rates are reduced as a result of increases in gas viscosity.
- Operating voltages are reduced primarily due to the lower densities of hot gases.
- Differential expansion may cause warping of structural components (44).

A significant advantage of electrostatic precipitation is that relative to other high efficiency collectors, ESPs exhibit low operating pressure drops -- 745 Pa (0.11 psi) -- and hence lower power requirements (45).

Fabric Filters -- In fabric filtration systems, dust-laden gas passes through a fabric in such a manner that dust particles are retained on the upstream side of the fabric, and clean gas passes through. As filtration proceeds, a filter cake builds up on the fabric which enhances particulate collection while increasing the operating pressure drop (45). Periodic

cleaning of the fabric filter is required to prevent excessive pressure drops. Three different cleaning methods are in use:

- reverse air cleaning, the gentlest method, which entails the reverse flow of gas through the filter to dislodge collected material;
- mechanical shakers, which displace filter bags in either the horizontal or vertical direction to dislodge collected materials; and
- pulse-jet cleaning, the most severe cleaning method, in which expansion of the bag, due to the pulsing of pressurized air into the bag, causes collected material to be dislodged.

Most utility fabric filters are cleaned by the reverse air method, although shaking is also commonly employed, but to a lesser extent (46).

Filter fabrics are normally woven with relatively large open spaces [on the order of  $1 \times 10^{-4}$  m (0.004 in) or larger]. Since particulates of much smaller size can be readily collected by these devices, the filtering process must involve mechanisms other than simple sieving. These mechanisms include interception, impingement, diffusion, gravitational settling and electrostatic attraction. Once a cake of dust is formed on the filter, further collection is accomplished by these mechanisms (in addition to sieving)(47).

Fabric filters employed in utility applications exhibit high collection efficiency; in a number of cases, actual operating efficiencies exceeded design collection capacities which generally ranged from 99.4 to 99.7 percent (46). Unlike ESPs, the effectiveness of fabric filters is independent of coal composition. Consequently, these devices can collect the high electric resistivity fly ash (i.e., that generated as a result of burning low sulfur coal) that is not amenable to collection by electrostatic precipitation. In addition, with the recent commercialization of dry FGD scrubbing systems (through the use of spray drying or dry injection of adsorbents into the flue gas) and the resultant production of essentially moisture-free solid wastes, the use of fabric filters may increase. Another advantage of fabric filters is their high efficiency in collecting submicron particles. However, fabric filtration has been shown to be more energy intensive than electrostatic precipitation. Additionally, close control of flue gas temperatures is required to prevent acid condensation which would result in bag deterioration and shortened bag life.

Wet Scrubbing -- Wet scrubbing for particulate control involves the removal of dust from a gas stream by absorbing of the particles into suspended liquid droplets or by adhesion of the particles to the scrubber walls followed by liquid flushing into a waste disposal system (48). Intimate contact between the gas stream and liquid is required to transfer suspended particulate matter from gas to the liquid. Dust removal efficiency of scrubbers is primarily a function of the energy consumed in contacting the dust and liquid.



The primary advantage of wet scrubbing systems as air pollution control devices on coal-fired boilers is their ability to collect both particulate materials (fly ash) and gases (sulfur oxides). The most common application of scrubbers for coal-fired boilers is for the simultaneous control of particulate matter and sulfur oxides. However, a number of disadvantages are also associated with the use of scrubbers for particulate control:

- Compared to other particulate control systems, high energy penalties (high-pressure drops) are associated with the use of scrubbers [about 1240 Pa (0.18 psi) for fabric filters compared to 6220 Pa (0.90 psi) for venturi scrubbers].
- Scrubbers are not as efficient as other particulate control systems with regard to the collection of fine particles [less than  $10^{-4}$  m (0.004 in)].
- Corrosion may potentially be associated with wet collection systems.
- A slurry waste is generated which may result in the need for relatively more expensive disposal operations (48).

#### 6.1.2.2 Flue Gas Desulfurization Technology--

A wide variety of FGD processes have been developed for application on utility boilers. As discussed in Section 3.2, the technology can be grouped into two categories: non-recovery (or throwaway) systems, which produce waste material for disposal, and recovery systems, which produce salable byproducts. Nonrecovery processes make up the overwhelming majority of the technology.

There are two types of non-recovery FGD processes: (1) wet processes which involve contacting flue gas with aqueous slurries or solutions of absorbents to produce wastes in the form of solutions or slurries; and (2) dry processes which produce essentially moisture-free solid wastes through the use of spray dryers or dry adsorbent injection.

Nonrecovery Wet FGD Processes -- Direct lime and/or limestone scrubbing systems constitute the majority of operational wet FGD systems. In these systems, the flue gas is contacted with a slurry of calcium salts (calcium sulfite/sulfate and calcium hydroxide or calcium carbonate) at total suspended solids concentrations of 8 to 16 weight percent. Slurries are recirculated through open venturies or spray-tower scrubbers at high liquid-to-gas ratios. The spent liquor is collected in a delay tank to allow for completion of the precipitation reactions. Fresh alkali makeup (either slaked lime or a slurry of finely ground limestone) is added to the delay tank, and a slipstream is removed for waste solids separation. The rate of fresh alkali makeup is controlled on pH (lime systems usually operate at a pH of 6 to 7; the pH of limestone is typically 5 to 6).

There are a number of variations on conventional direct lime and limestone scrubbing systems directed toward improving sulfur oxides removal and overall scrubber system reliability. Many full-scale lime scrubbing

systems, for example, use either carbide lime or a dolomitic (high magnesium content) lime rather than commercial lime. (Dravo Corporation markets a special high magnesium content lime for FGD systems under the trade name, Thiosorbic lime.) The use of these limes effects lower oxidation rates and, therefore, better control of scrubber scaling. Also, the use of high magnesium content lime or the addition of magnesium oxide to commercial lime usually results in higher sulfur oxides removal efficiency and improves lime utilization for high sulfur coal systems (reducing lime stoichiometries from typical levels of 1.1-1.3 to 1.0-1.15).

A principal variation on direct limestone scrubbing involves intentional oxidation (i.e., forced oxidation) of the calcium sulfite salts formed in the scrubber to calcium sulfate. This is done to improve sulfur oxides removal efficiency, minimize scale and plugging potential, improve solids dewatering properties (by converting the wastes to gypsum), and increase limestone utilization. It is hoped that use of forced oxidation in high sulfur coal applications will reduce limestone stoichiometries from the typical levels of 1.25-1.5 to 1.05-1.1. In the simplest form of the forced oxidation process, air is bubbled through the slurry in a modified delay tank; however, two-stage scrubbing has also been used to cause intentional oxidation.

Western United States coal reserves include mostly lignites and subbituminous coals which contain far less sulfur than typical Eastern coals. Another feature of virtually all Western coals is that their ashes have high contents of calcium, sodium, and magnesium oxides. This lower required sulfur oxides removal combined with the alkalinity of the fly ash has led to the use of fly ash slurries as scrubbing liquors in several Western power plants. Fly ash alkali (FAA) scrubbing typically involves process configurations similar to lime or limestone scrubbing. The chemistry of FAA systems is much more complex due to the greater number of competing reactions. Good utilization of the ash alkali also necessitates operation at lower pH levels (3.5 to 5.0) than limestone systems. Supplemental alkali (typically lime or limestone) is frequently required.

Another wet, non-recovery FGD process is the dual alkali process. In dual alkali systems, sulfur oxides removal is accomplished with solutions of sodium salts which are then regenerated with lime to produce a waste calcium-sulfur salt similar in chemical composition to the waste produced from direct scrubbing systems. Dual alkali systems are most appropriate for medium and high sulfur coal applications where relatively high sulfur oxides removal efficiencies are required and where oxidation rates tend to be relatively low. Filters are used to recover sodium salts in order to minimize sodium carbonate makeup requirements and reduce the potential adverse environmental impacts from the high total dissolved solids (TDS) levels in the waste material.

Wet scrubbing processes can also entail the use of sodium salts, with no regeneration. However, this technology is now used principally for industrial boiler applications rather than coal-fired utilities. In this case, a liquid waste stream is produced. The high cost of the sodium reagent and the problem of liquid waste disposal have significantly limited utility

applications. The few utility applications use inexpensive sources of alkaline sodium salts, such as trona. Once-through sodium systems are capable of achieving very high sulfur oxides removal efficiencies, approaching 99 percent. The type of scrubber is dependent primarily upon the degree of sulfur oxides control required. The pH is controlled in the range of 5 to 7 through the addition of fresh alkali, and a slipstream of spent liquor is removed for discharge to lined holding ponds.

Nonrecovery Dry Flue Gas Desulfurization Processes -- As discussed in Section 3.2.3, three different approaches to dry scrubbing for sulfur oxides control and particulate collection have been actively pursued, although only one--the injection of solid sorbents into the flue gas stream with collection downstream in a particulate control device--has reached commercialization. In this commercial practice, an aqueous solution of slurry sorbent is injected as a fine mist into the ash-laden flue gas in a spray chamber. The hot gas evaporates the water, and some sulfur oxides are removed through reaction with the alkali. The gas then passes to a dry particulate collector (fabric filter or electrostatic precipitator) where the fly ash and dry sorbent solids are removed and further sulfur oxides removal is achieved. The flue gas is exhausted directly to the stack without reheat, since the flue gas is not usually allowed to reach saturation in the spray dryer.

The remaining two approaches to dry FGD have not reached commercialization. Testing of dry sorbent injection has focused on the use of sodium bicarbonate nahcolite, a material not available in commercial quantities. Injection of sorbents into the boiler combustion zone is also in the testing phase.

Recovery FGD Processes -- Recovery FGD processes can also be categorized as wet and dry, according to the mode of sulfur oxides removal. As noted in Section 3.2.2, they can be further classified according to the type of byproduct produced, i.e., concentrated sulfur oxides for conversion to sulfur or sulfuric acid, sulfur only, or acid only.

Recovery FGD systems produce relatively small amounts of waste compared to non-recovery FGD systems and comprise a minor portion of the total FGD system installations or applications. The Wellman-Lord and magnesium oxide processes are the two major recovery FGD processes that have been commercially applied by utilities.

The Wellman-Lord process uses a sodium sulfite solution to absorb sulfur oxides, forming a solution of sodium bisulfite. Makeup sodium carbonate also reacts with sulfur oxides in the absorber, producing sodium sulfite. Side reactions produce some sodium sulfate, which is recovered by treatment of a purge stream. Sodium sulfite is regenerated from the bisulfite by simple addition of heat, liberating sulfur dioxide gas, which is recovered and either reduced to elemental sulfur or used to make sulfuric acid.

Magnesium oxide scrubbing of sulfur oxides produces magnesium sulfite/sulfate salts which are recovered and dried. The magnesium oxide is

then regenerated in a calciner, liberating sulfur dioxide, which is again recovered and either reduced to sulfur or used to make acid.

#### 6.1.3 Coal-Fired Utility Solid Waste Handling and Processing

Waste handling and processing options for coal-fired utility wastes vary with both waste types and the physical state(s) of the wastes. This discussion of coal-fired utility waste handling and processing is segmented by waste type, since, for each waste type, a variety of waste handling and processing alternatives is available.

##### 6.1.3.1 Fly Ash Handling and Processing--

The selection of fly ash handling and processing systems is dictated, to a large extent, by the following factors:

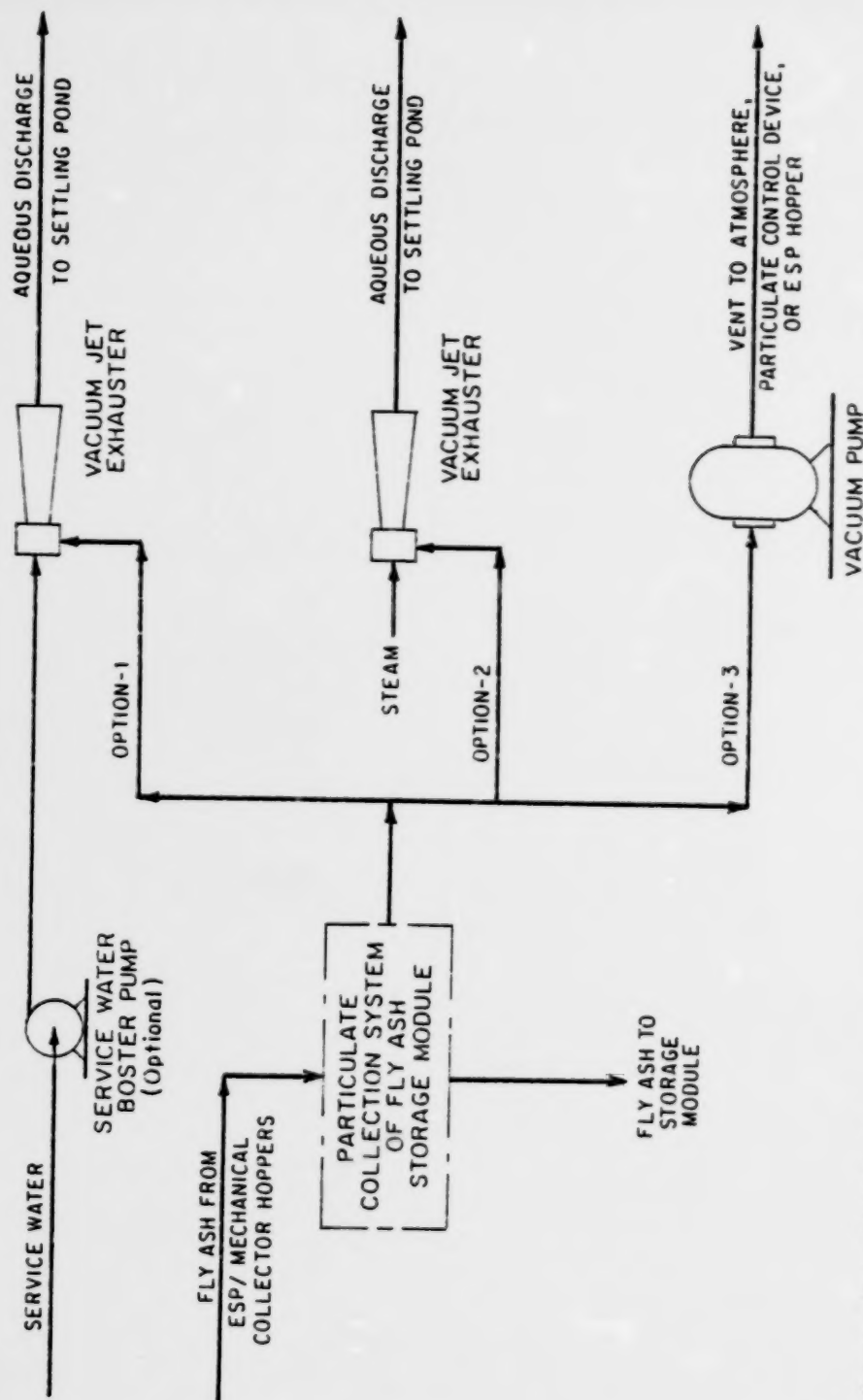
- the form of the waste available to the handling and processing system (e.g., fly ash is collected in dry form by ESPs and mechanical collectors, while aqueous fly ash slurries are generated by flue gas scrubbers);
- the need to handle and process fly ash wastes due to the presence of other waste species (e.g., scrubbers designed to simultaneously collect particulates and sulfur oxides yield aqueous slurries of fly ash and FGD wastes; the presence of FGD wastes may call for dewatering of the combined wastes prior to disposal); and
- the ultimate fate of the fly ash (i.e., utilization or disposal by wet ponding or dry impoundment).

In cases where fly ash is collected in dry form, the method of waste handling and processing is primarily dependent upon whether the waste will ultimately be utilized, disposed in dry impoundments, or disposed in wet ponding operations. Handling and processing operations for fly ash which will be utilized or disposed in dry form are essentially the same. In these cases, dry fly ash is conveyed pneumatically from the collection device to storage silos for intermittent transfer for disposal. For short conveying distances of 185 m (55 ft) or less, vacuum conveying is typically used. Vacuum system motive force can be produced either hydraulically or mechanically by any of three devices:

- a jet exhauster using water to create a vacuum;
- a jet exhauster using steam to create a vacuum;
- a mechanical vacuum pump which operates on electrical energy to create vacuum.

Figure 6.3 illustrates the various options available in vacuum pneumatic conveyors for fly ash handling.





Source : Arthur D Little, Inc.

FIGURE 6.3 FLY ASH HANDLING AND PROCESSING MODULE  
Vacuum Pneumatic Conveying Options

For longer conveying distances, pressure conveying is the most economical method. Air locks are used to transfer fly ash from the hoppers of the air pollution control devices to the transport lines. A positive displacement blower is used to create the motive force. This system is shown schematically in Figure 6.4.

It is sometimes economical to employ a combination of vacuum and pressure conveying of fly ash where distances rule out the use of vacuum conveying alone. In this manner, the advantages of both systems are utilized. The vacuum system, with its simplified controls, removes the fly ash from the collection device hoppers at an optimum rate and transports it to a single transfer point. From this point, the pressure system economically delivers the fly ash to the terminal destination. The combination vacuum/pressure pneumatic fly ash handling system is shown in Figure 6.5.

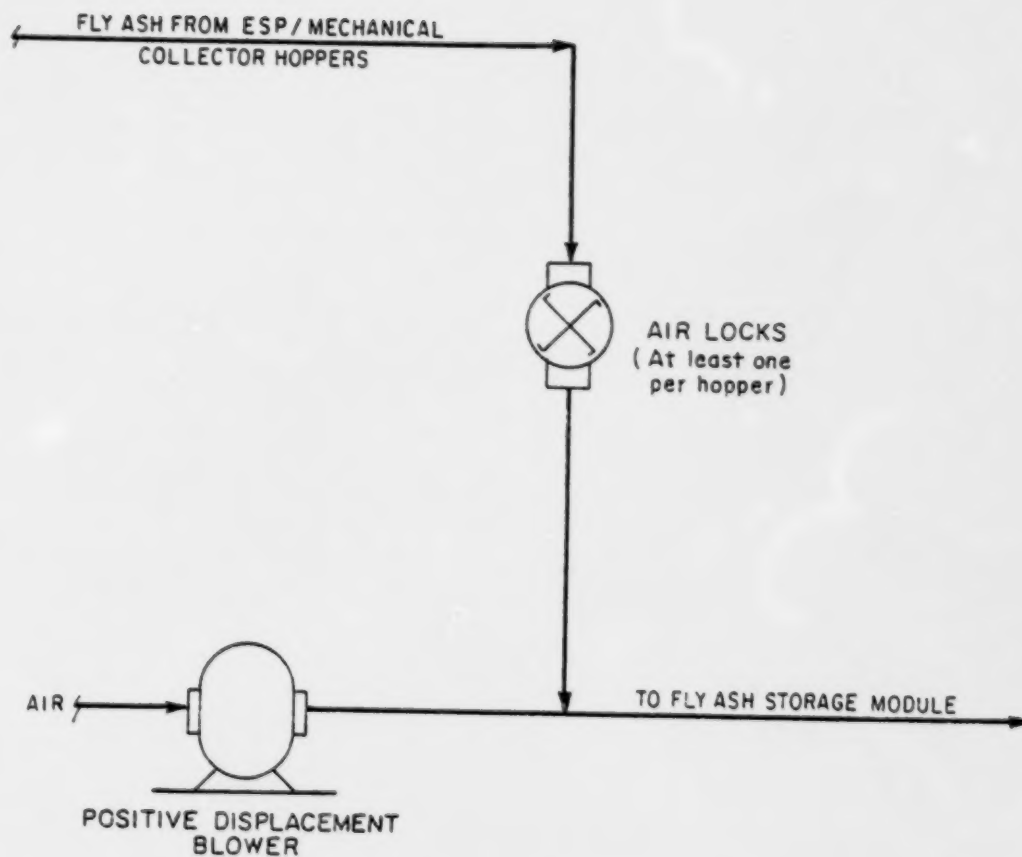
When ponding is the disposal method of choice for fly ash that is collected dry, the ash may be directly sluiced from the collector hopper with a hydro-ejector system and then pumped to the pond. An alternate system involves a vacuum pneumatic system to carry the ash from the hoppers to a water jet exhaustor which is used to create the vacuum; the vacuum-producing water is used to slurry the fly ash. The slurry is discharged through an air separator tank before being transported to the disposal pond. This type of system provides for continuous fly ash removal. In some cases, provision is made for returning supernatant back to the ash sluicing system. Figure 6.6 illustrates options for wet handling of fly ash collected in dry form.

In the very limited number of cases where fly ash is collected in wet form (via wet particulate scrubbers), it is usually pumped to a pond in slurry form. The wet ash may, in some instances (typically when collected simultaneously with FGD waste), be pumped to a thickener or clarifier which partially dewateres the FGD waste and the accompanying ash.

Dewatering of fly ash wastes may also be achieved by interim ponding. The aqueous fly ash slurry is placed in the pond and the liquor decanted or evaporated. Once sufficient dewatering has occurred, the waste is dredged and disposed in a landfilling operation. Typically, two ponds are used. One pond is kept in service while the second is being dewatered and dredged. Interim ponds are similar to disposal ponds in construction. (See Section 6.1.7.2.)

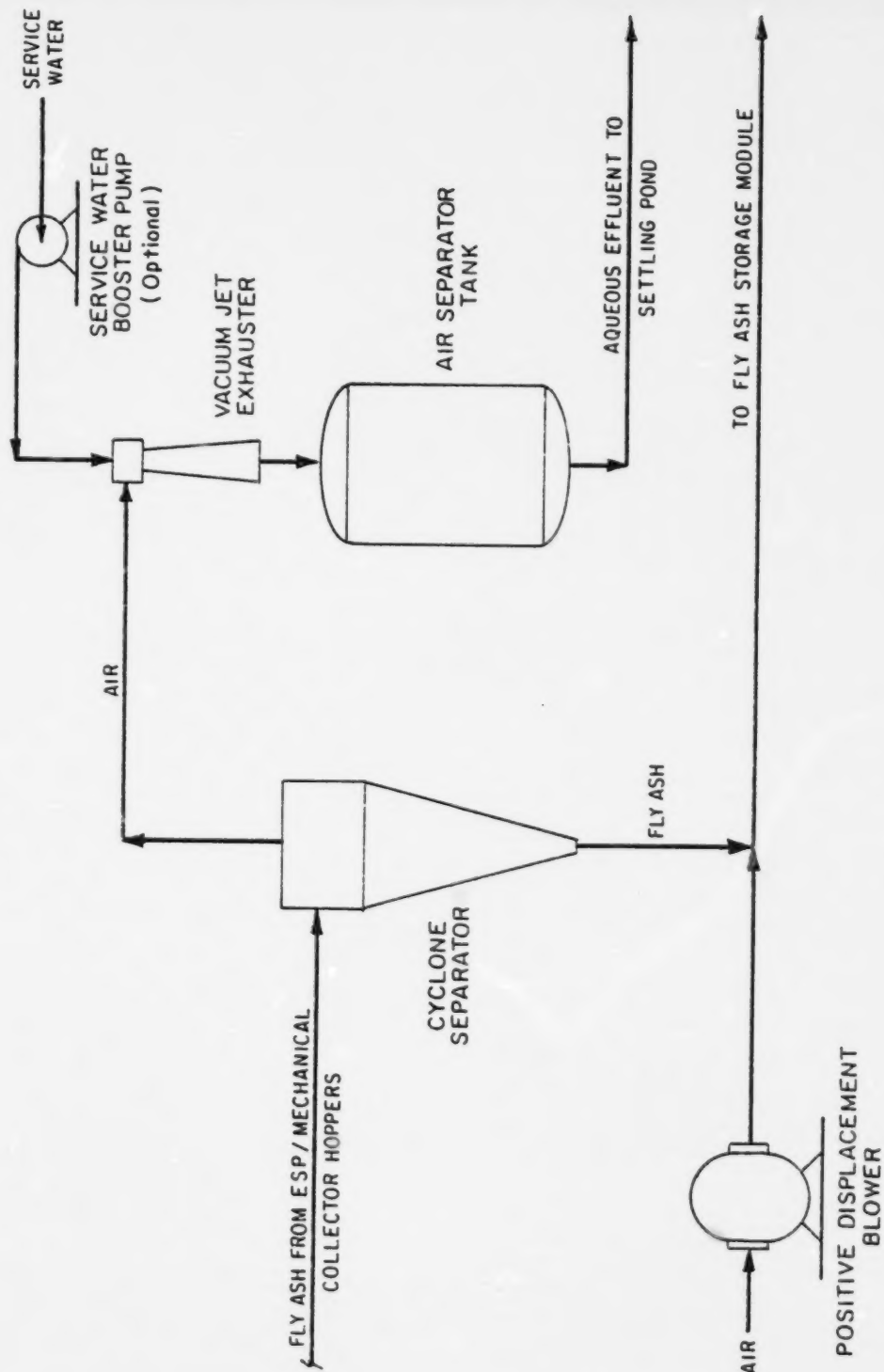
If the interim ponded waste has been sufficiently dewatered to bear the weight of conventional excavation equipment, various types of dozers and shovels may be used for the dredging operation. If the dewatered waste cannot safely support this type of operation, dozers and shovels may be required to work from the perimeter of the pond or from dikes and proceed into the pond as waste is removed. Even as the waste is removed, dewatered bottom ash is sometimes employed as a base road building material to allow dozers and/or scrapers to enter onto the pond.





Source: Arthur D. Little, Inc.

**FIGURE 6.4 FLY ASH HANDLING AND PROCESSING MODULE**  
**Pressure Pneumatic Conveying System**



Source : Arthur D. Little, Inc.

**FIGURE 6.5 FLY ASH HANDLING AND PROCESSING MODULE**  
Combination Vacuum/Pressure Pneumatic Conveying System

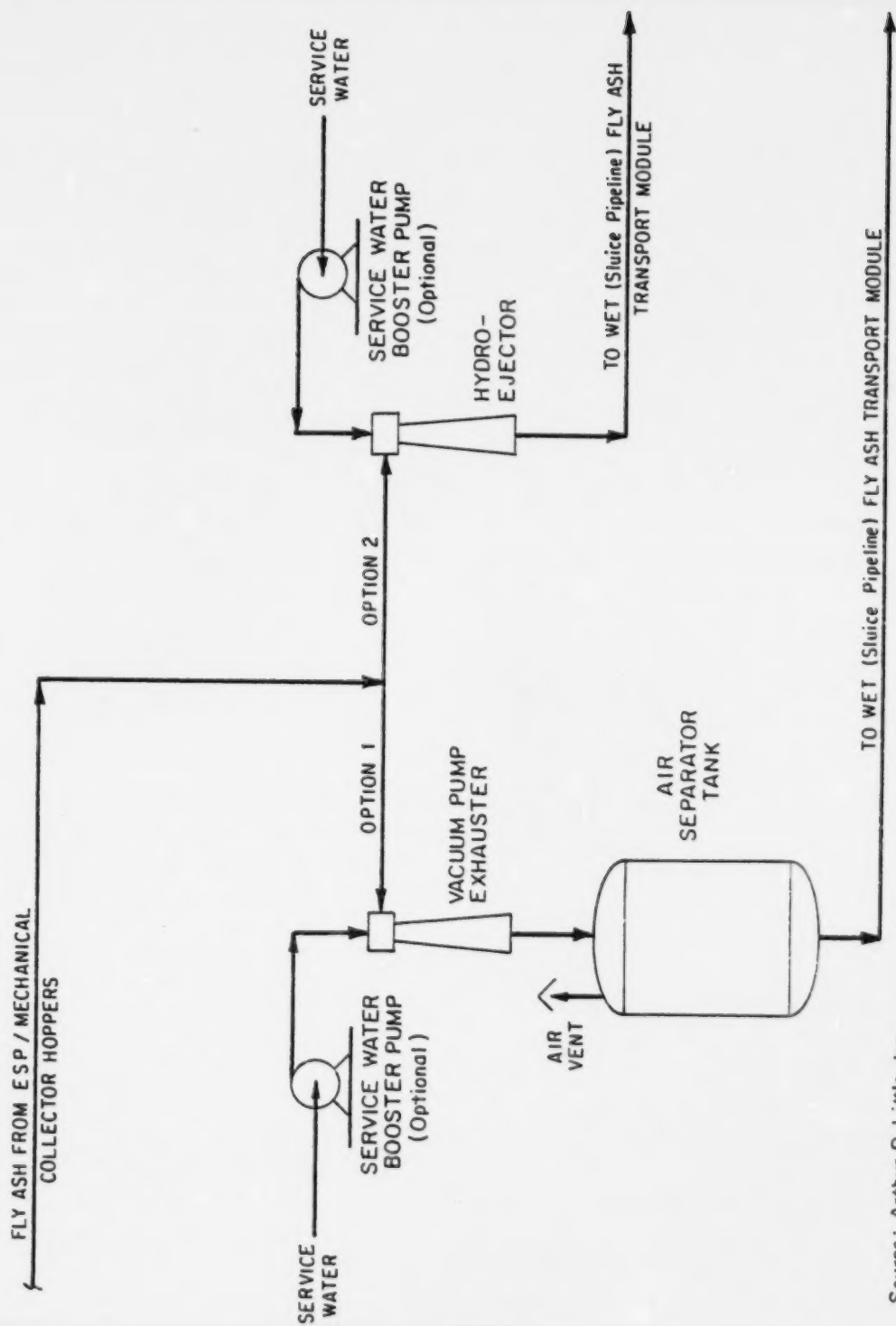


**FIGURE 6.6 FLY ASH HANDLING AND PROCESSING MODULE**  
Combination Vacuum Pneumatic Conveying – Wet Sluicing System

If the waste is not sufficiently dewatered to permit this type of excavation or if the pond is too large to be excavated by shovels, a cable powered dragline may be employed. Draglines have been in use for more than 50 years for long range digging and hauling, with scraper sizes ranging from 0.4 to 15 m<sup>3</sup> (0.5 to 20 yd<sup>3</sup>) and on spans of up to 305 m (1,000 ft). Skid-mounted units are available in sizes up to 2.3 m<sup>3</sup> (3 yd<sup>3</sup>). These machines offer high production and portability where the line of operation must be frequently shifted (Figure 6.7).

Draglines work equally well on high banks [measuring as high as 73 m (240 ft)], level ground, and under water. Hauls from the deposit can be made to a surge pile or hopper. Dragline machines are built in sizes to match all tonnage requirements. Both the size of the dragline and the length of haul are the chief factors in determining handling capacity.

The drag line consists of a bottomless scraper, operating cables, a two- or three-drum hoist, and guide blocks. The blocks may be mounted on a mast, skid frame, or mobile towers located at the head and tail ends of the installation. The cables are spooled in the drums, reeved through the guide blocks, and drag the scraper through its operating cycle. Scrapers can discharge their load at the head end support into a surge pile, or into a hopper, truck or rail car. With larger size scrapers, 4 m<sup>3</sup> (5 yd<sup>3</sup>) and larger, a track cable is used to lift the scraper off the surface on the return cycle and to lower the scraper to contact the waste material during the digging and waste hauling cycle.

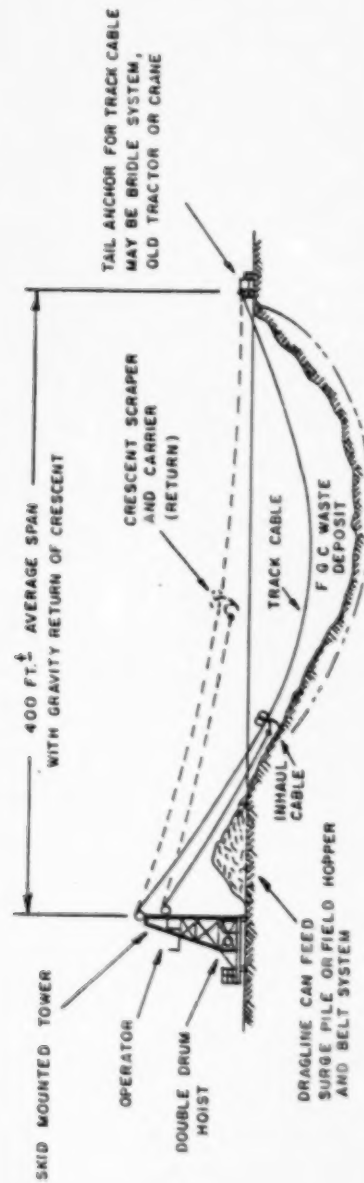
In some interim ponding or disposal ponding operations, supernatant recycle may be practiced. This operation is similar to that discussed in the following section on FGD waste processing and handling.

#### 6.1.3.2 FGD Waste Processing and Handling--

FGD wastes, like fly ash, can be collected in either wet or dry form by currently available sulfur oxides control devices. Dry FGD waste technologies are relatively new and, therefore, most FGD waste handling and processing options in use are based on wet wastes. However, the use of dry sulfur oxide scrubbing technologies is growing at a rapid pace, and the handling and processing systems for wastes from these technologies warrant attention. In general, handling systems for dry FGD wastes would be similar to the dry fly ash handling systems discussed in Section 6.1.3.1.

There are four basic approaches to processing and handling wet FGD wastes:

- pumping the wet scrubber effluent directly to a disposal pond with no processing;
- dewatering scrubber sludge prior to pumping it to a disposal pond;
- dewatering scrubber sludge prior to stabilization of the dewatered waste by mixing with fly ash; and



SOURCE: Arthur D. Little, Inc.

FIGURE 6.7 SKID-MOUNTED DRAGLINE MACHINE



- dewatering of the FGD waste and subsequent chemical fixation prior to disposal.

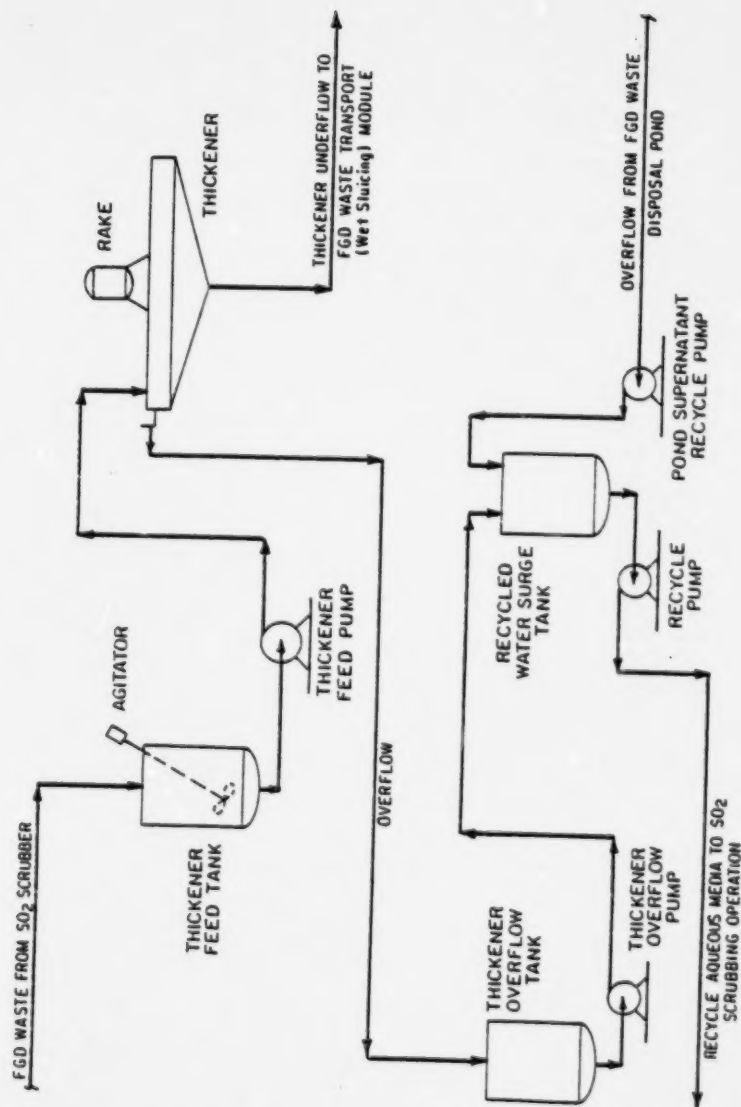
In cases where FGD wastes are directly ponded, the supernatant is generally returned to the process. This practice may also be employed when FGD wastes are dewatered prior to disposal. However, not all ponding systems incorporate water recycle, and many which do include this feature practice only partial recycle. As a rule, water management and conservation dictate water reuse. For new systems, recycle and optimum water management are anticipated to be more widely practiced.

Most unthickened slurry wastes produced by FGC systems contain on the order of 5 to 15 weight percent suspended solids. In order to avoid the unnecessary discharge of large amounts of process liquor, these wastes are frequently mechanically dewatered prior to being disposed. Dewatering decreases the volume of waste for transport and disposal, thereby reducing the costs of these operations. Additionally, wastes dewatered to an optimum moisture content exhibit a maximum strength; thus, dewatering to reach that moisture content is a useful engineering technique. Primary dewatering is usually accomplished with thickeners or interim ponds. Primary dewatering is virtually universally practiced to reduce sludge volume and conserve water. Figure 6.8 illustrates a typical primary thickening operation for pond disposal of FGD waste. Interim ponding of FGD wastes is practiced in the same way as interim fly ash ponding (Section 6.1.3.1). Secondary methods of dewatering are also sometimes used. These include vacuum filtration and centrifugation. Secondary dewatering is only practiced as a precursor to dry impoundment to improve the handling properties of the wastes prior to truck transport, stabilization or fixation. Figure 6.9 shows typical primary and secondary dewatering options.

Conventional gravity thickeners operate similarly to settling tanks. Feed solutions enter at the center and are distributed radially. A circulating rake is used to gently push settled solids toward the discharge point. Sludge solids are collected as underflow in a sump and then directed to disposal or secondary dewatering operations (i.e., vacuum filtration or centrifugation). Clarified aqueous effluent is discharged over a weir and is generally recycled to the scrubber system.

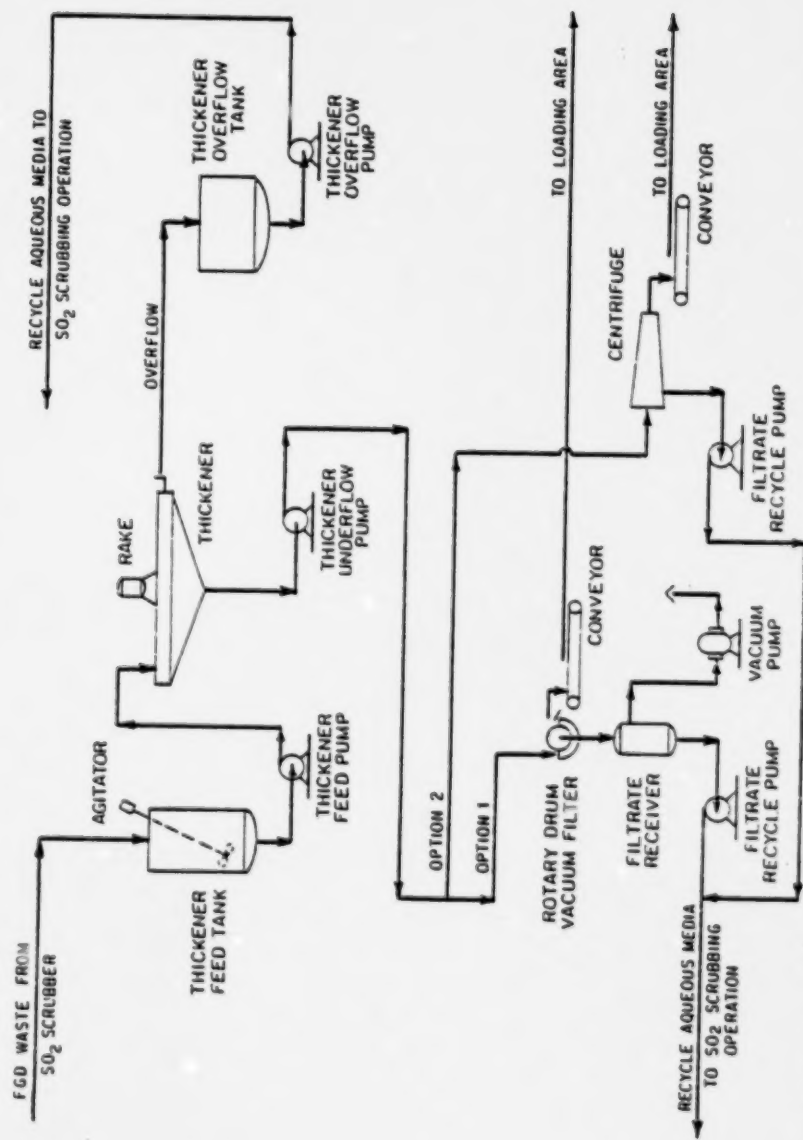
Two types of vacuum filters have been used to dewater FGD sludges, rotary-drum filters and belt filters. These filters are similar in operation, although a scraper is required to remove solids from the rotary drum filter cloth, while none is needed for the belt filter. From a maintenance standpoint, this scraping results in costly wear of the rotary drum filter cloth. However, the installed capital cost of belt filters can be up to 30 percent higher than that of drum filters. Both filters require auxiliary equipment such as a vacuum receiver, vacuum pump, and filtrate pump.

Two types of centrifuges are applicable to dewatering FGD scrubber sludges. These are the countercurrent centrifuge (in which solids are discharged at the slurry inlet end of the bowl) and the concurrent centrifuge



Source: Arthur D. Little, Inc.

FIGURE 6.8 FGD WASTE HANDLING AND PROCESSING MODULE  
Primary Dewatering System



Source: Arthur D. Little, Inc.

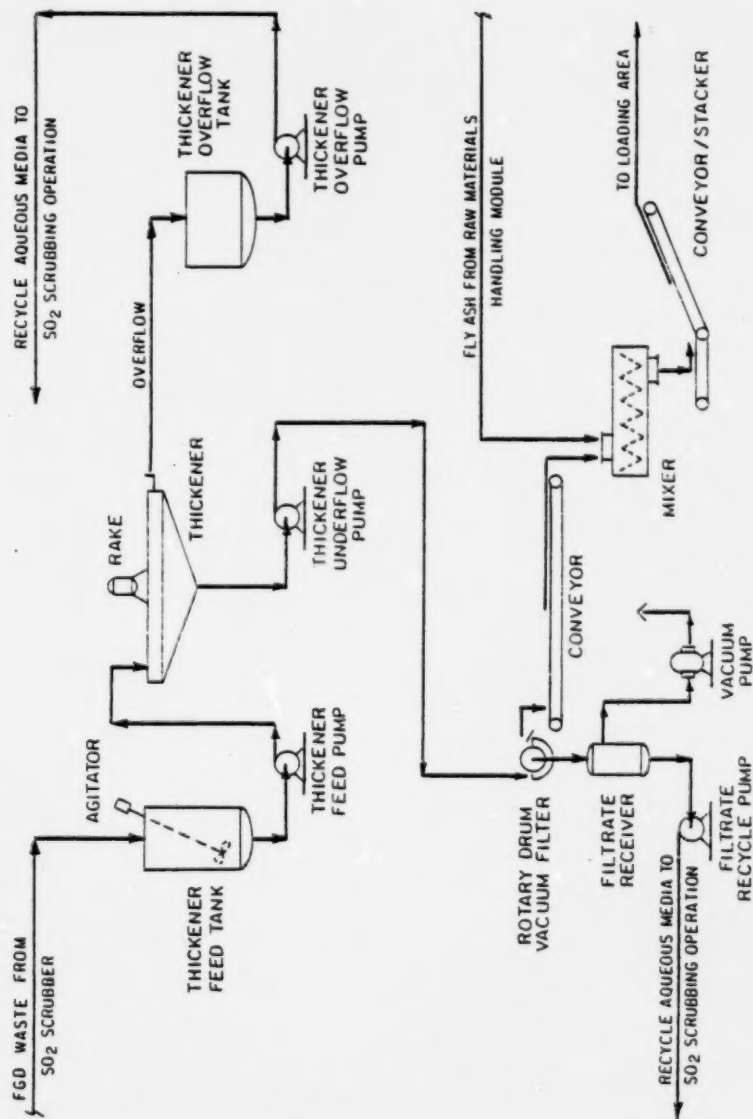
FIGURE 6.9 FGD WASTE HANDLING AND PROCESSING MODULE  
Primary and Secondary Dewatering Options

(in which solids discharge from the end opposite the slurry inlet). These centrifuges differ significantly. Concurrent centrifuges are large in size and operate at low centrifugal force, while countercurrent centrifuges are smaller in size but employ high centrifugal force. To date, the advantage of one type over the other for this application has not been proven.

Dry blending of FGD waste and fly ash (i.e., stabilization of FGD waste with fly ash) results in a third FGD handling and processing variation. In blending applications, FGD wastes are dewatered in primary and secondary operations, and the resulting waste is mixed in a pug mill with dry fly ash to produce a mixture with a lower moisture content than the FGD waste alone. Figure 6.10 is a process flow diagram of FGD waste/fly ash blending. A variation on this process is to mix the fly ash with the FGD sludge at the landfill site with earth moving equipment. Control of moisture content to obtain the optimum compaction in the landfill is essential to this process. Fly ash reactivity is a second parameter of importance, since use of alkaline fly ash containing significant quantities of calcium oxide can cause the co-disposed waste to undergo a pozzolanic reaction, thereby generating a fixated waste. This type of reaction can be desirable; in fact, processes based on other additives have been designed to produce fixated wastes, as described below.

The fourth FGD waste handling and processing option is chemical fixation. In this process, fly ash and lime or other additives are added to dewatered FGD wastes prior to disposal. In the fixation process, the FGD wastes and cementitious additives undergo pozzolanic reactions that produce increased structural integrity of the resulting wastes. This chemical process continues even after placement of the process waste.

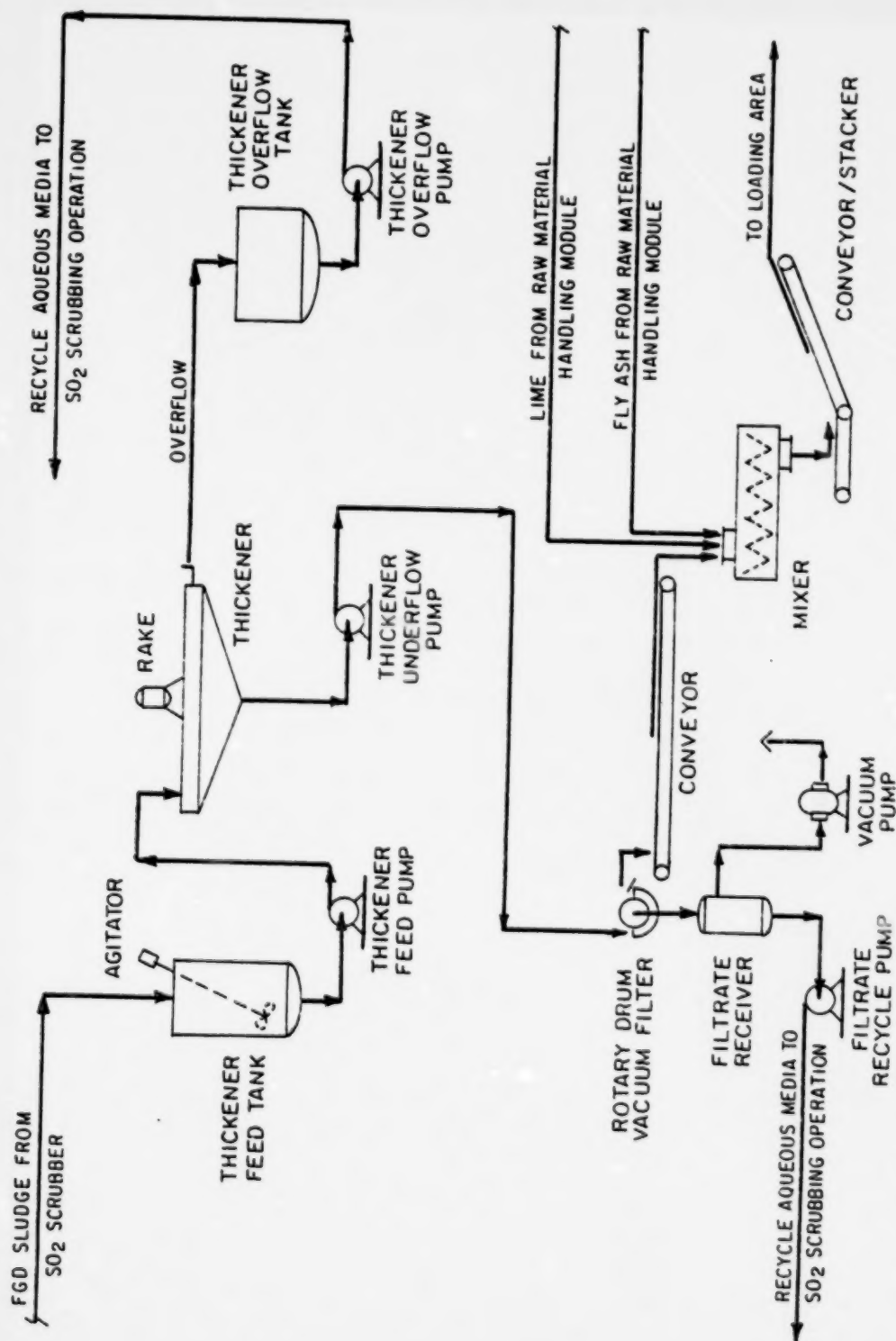
The most common FGD waste fixation process in use today is the Conversion Systems, Inc. (CSI) process. In it, the FGD wastes undergo primary and secondary dewatering to a high solids content (50 to 60 percent). A thickener and filter are typically employed. Subsequently, the waste is blended with fly ash and lime in a pug mill, where the pozzolanic reaction yields a material with cementitious properties. [The raw materials (i.e., fly ash and lime) storage and delivery system is described in Section 6.1.5.] The fixated waste that leaves the mixer is transported by belt conveyors to temporary surge piles for storage until it is reclaimed for ultimate disposal. Single stationary stacking conveyors are satisfactory for delivering dry sludge to surge piles of moderate size. However, only conical piles can be built with a stationary conveyor and because of the height limitations of such piles, it is often necessary to distribute the surge pile over a larger area than is possible with a stationary conveyor. Therefore, many utilities use inclined conveyors pivoted at the tail end on the loading point centerline (i.e., radial stackers). The conveyor structure is supported near the midpoint by a tower. The radial stacker design allows for movement of the conveyor discharge point, thereby permitting the formation of large capacity arc-shaped piles. Figure 6.11 is a process flow diagram of the CSI fixation process.



Source: Arthur D. Little, Inc.

FIGURE 6.10 FGD WASTE HANDLING AND PROCESSING MODULE  
FGD Waste/Fly Ash Blending System





Source: Arthur D. Little, Inc.

FIGURE 6.11 FGD WASTE HANDLING AND PROCESSING MODULE  
CSI Fixation Process

A second chemical fixation process of interest is the Dravo process. The Dravo Lime Company patented a method of FGD waste fixation through the use of hydrated lime and Calcilox®, a cementitious product derived from basic blast furnace slag. Dravo offers three distinct variations of its process:

- direct ponding;
- interim ponding followed by landfilling; and
- direct landfilling.

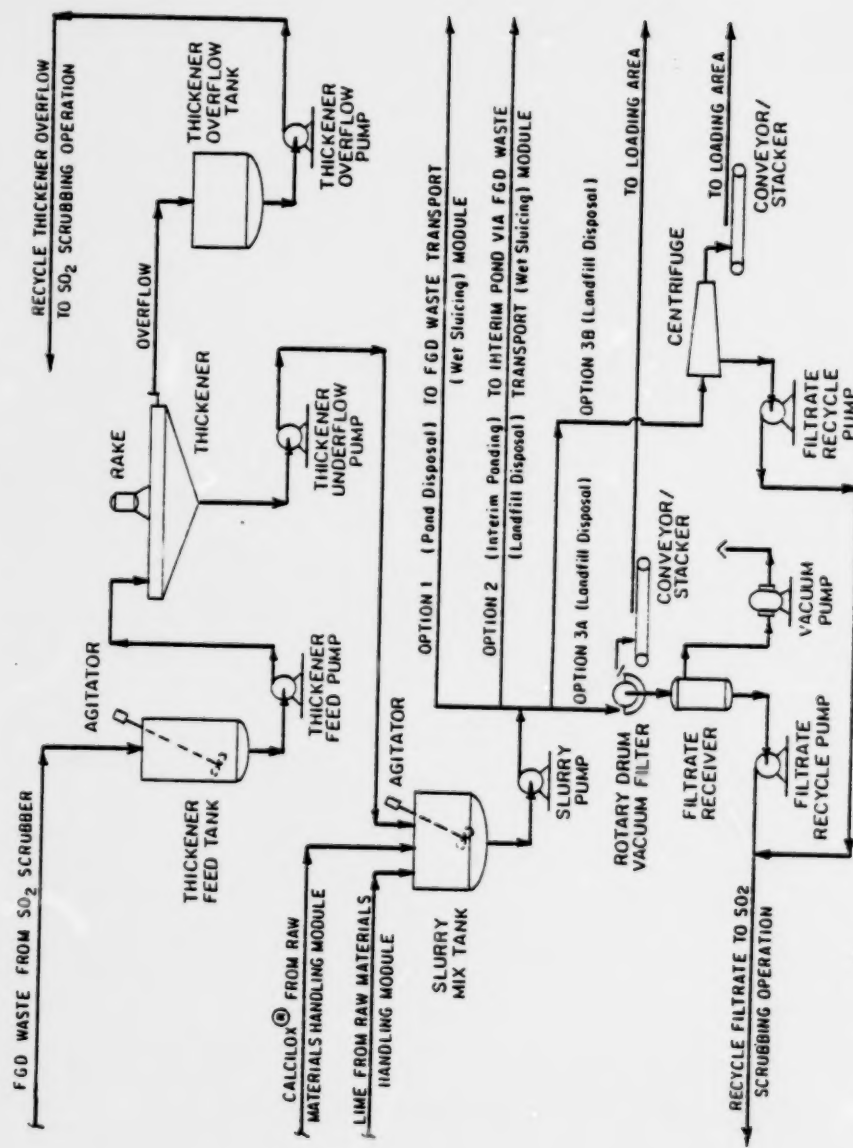
In the Dravo ponding process, FGD waste is dewatered in a thickener to 25 to 30 weight percent solids. The dewatered waste is mixed in a tank for 1,200 sec (20 minutes) with the lime and Calcilox fixation agents. Normally 5 to 10 percent of Calcilox is added in terms of dry sludge solids weight. The lime is added in sufficient quantity to raise the pH of the mixture to between 10.5 and 11.0. After mixing, the treated waste slurry is pumped to the disposal pond. It is in the pond that the fixated waste solidifies to form an earth-like substance called Synearth. The pond supernatant is usually returned to the scrubbing system, since the highly alkaline liquor cannot be discharged to local surface water streams without costly acid treatment.

The remaining two Dravo processes employ much of the same FGD waste processing methods that are used by Dravo to treat wastes for pond disposal. With the interim pond option, the major difference is that the treated waste is dewatered in the pond and is dredged and ultimately disposed in a landfill. The direct landfilling option calls for dewatering of the treated waste slurry from the mixing tank by the secondary methods (e.g., filtration and centrifugation) previously described. Figure 6.12 illustrates the various Dravo options for FGD waste handling and processing.

Forced oxidation is a waste processing step which is incorporated as part of the scrubber cycle. Here sulfite species are intentionally oxidized to sulfate so that gypsum (which is more readily dewatered) is the primary FGD sludge component. Forced oxidation involves addition of air to the scrubbing liquid or recycle loop bleed stream by means of compressors or an eductor. The control of pH is essential, since the oxidation rate increases as pH decreases. The pH can be controlled by employing a two-stage scrubbing process, in which the first stage is maintained at low pH for oxidation purposes and the second stage at a higher pH for sulfur dioxide removal. The ultimate value of forced oxidation system is with respect to the nature of the waste generated. Gypsum is readily dewatered, unlike calcium sulfite wastes; thus, forced oxidation systems reduce FGD waste processing costs as well as those for transport and disposal.

#### 6.1.3.3 Bottom Ash and Slag Handling and Processing--

Bottom ash and slag collection systems differ, depending upon the form of the waste; bottom ash may be collected in dry form from dry bottom boilers, while slag is collected when a wet bottom boiler is employed.



Source: Arthur D. Little, Inc.

FIGURE 6.12 FGD WASTE HANDLING AND PROCESSING MODULE  
Dravo Fixation Process

Bottom ash produced by dry bottom boilers is collected in hoppers. Different types of hoppers are used, depending upon the capacity of the boilers which they service. For smaller boilers, with steam generating capacities in the 126 kg/sec (1.0 million lb/hr) or smaller range, ash feeding is not a consideration, and flat-bottom ash hoppers with a single end or center outlet are used. Such ash hoppers provide maximum storage with limited headroom. Ash must be fed from the flat bottoms by means of operator-controlled nozzles. Larger boilers require hoppers with large storage capacities, steep self-feeding slopes, and water containment to quench ash and to provide delivery at a high rate to the conveying system. The bottom ash hopper will be equipped with refractory-lined ash gates, which may be water cooled if high temperatures are anticipated. These ash gates may be either manually operated or equipped with an air-operated cylinder.

For continuously slagging, wet-bottom boilers, slag tanks are employed. Ash leaves the boiler in a molten state. Slag is quenched in a water-filled hopper. The quenching medium may, in some instances, be used as the transport fluid for sluicing the quenched ash. The slag hopper design is similar for wet bottom pulverized coal-fired boilers and for cyclone boilers. However, larger hoppers will be required for the latter system, since the percentage of bottom ash recovered in cyclone furnaces is higher than in the pulverized coal-fired type of boiler.

Bottom ash and slag handling and processing requirements are dictated primarily by the disposal method. Handling and processing options include sluicing for bottom ash or slag disposed by wet ponding and dewatering for dry disposal. In either case, the bottom ash is generally passed through a grinder before it is handled or processed. Double and single roll clinker grinders are used. This crushing is sufficient to produce particles which will pass through ejectors, pumps, and conveying lines.

Bottom ash and slag are typically sluiced with jet pumps, although slurry pumps have been employed. With jet pumps, water provides the motive force. The waste may be sluiced to the disposal area for wet disposal or to a hydrobin for dewatering when dry disposal is called for. Hydrobins are available in a range of capacities to meet the requirements of plants of varying sizes. These capacities range from 23 to 900 metric tons (25 to 1000 tons), based on a dry coal ash density of  $720 \text{ kg/m}^3$  ( $45 \text{ lb/ft}^3$ ). In most cases, at least two hydrobins are employed. One bin is available to receive ash, while the second contains ash which is being dewatered and unloaded. Ash is pumped into the bin over a bar screen, which permits the finer material to drop directly into the center while the coarser particles are diverted toward the sides of the bin to form a filter to trap fines before they reach the decanting elements. Aqueous overflow enters a serrated weir around the bin periphery and subsequently flows into a trough to a drain or settling system. The flow over the weirs is reasonably steady and undisturbed due to the presence of an underflow baffle. All material entering the bin must pass under this baffle before reaching the weirs. In this manner, turbulence caused by the inlet discharge does not reach the weirs. Once a bin is full, it is allowed to stand for about 3600 sec (one hour) until most ash has settled. The standing water is drained by a

floating decanter. Water is drained from the body of the ash through stationary decanters on the sidewalls of the bin. Normally, about 28,800 sec (eight hours) are required for the total dewatering process, if a product satisfactory for truck or rail transport is desired. Ash is discharged through the bottom of the bin by means of a hydraulically operated gate.

Various bottom ash handling and processing options are shown in Figures 6.13 and 6.14.

#### 6.1.4 Coal-Fired Utility Solid Waste Storage

Until recently, dry fly ash was the only coal-fired utility waste which required provisions for its storage. With the advent of dry FGD scrubbing, the need arose for storage of wastes from this process also. Fly ash can be stored for intermittent removal for sale, for use in FGD waste stabilization/fixation, or for disposal. Dry FGD wastes will typically be stored prior to disposal.

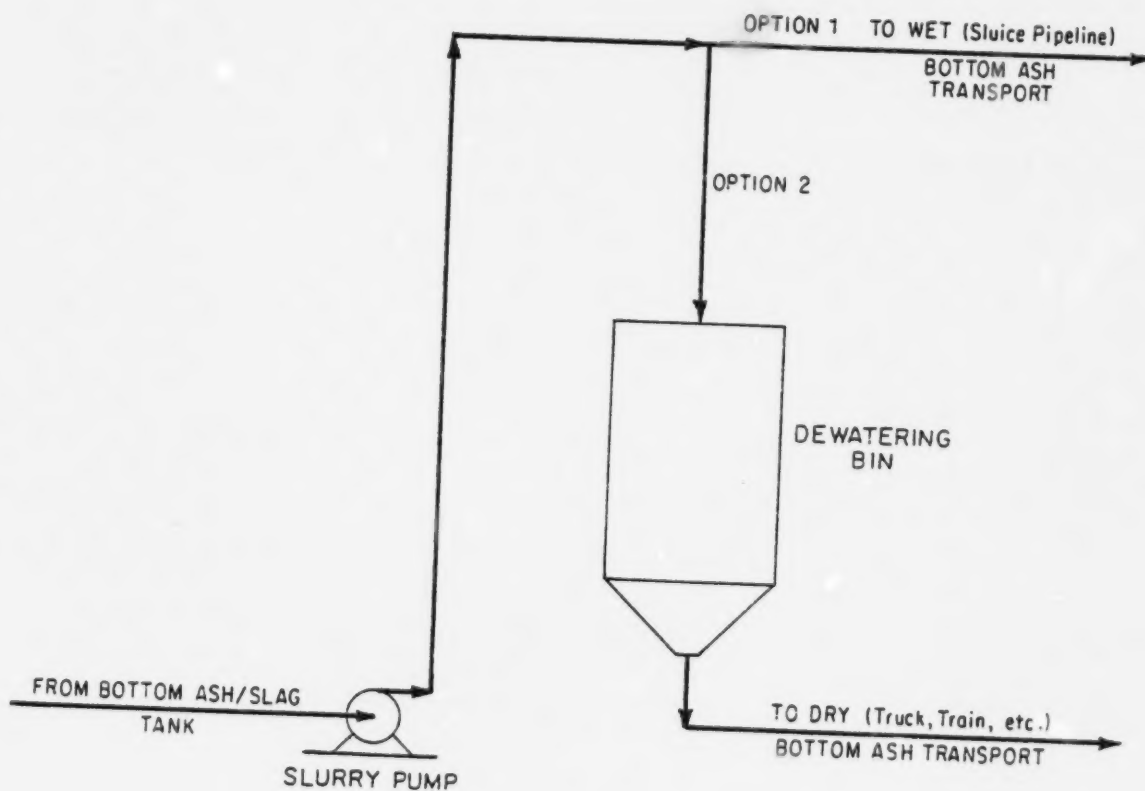
Dry fly ash is conveyed pneumatically (see Section 6.1.3.1) from the air pollution control device to the storage area. Fly ash can be separated from the conveying air by means of cyclone separators or fabric filters located on top of the storage silo. Collected fly ash is discharged through the bottom of these devices into the storage silo. Cyclone separators are inertial separators which remove particles from the carrying gas by transforming the velocity of inlet stream into a confined double vortex. The gas moves downward in spiral fashion at the outside of the cyclone and upward at the center to discharge at the top. The particles, due to their inertia, move toward the cyclone wall and are collected at the bottom of the cyclone.

Fabric filters provide higher efficiency than cyclones. The fly ash laden air is passed through a fabric bag which filters the ash from the gas. A number of these fabric filters are housed in a single baghouse. Fly ash is removed from the bags by shaking, pulsing a jet of air through the bags, or reversing the air flow through the bags.

Alternately, fly ash and the accompanying conveying air may be discharged directly into the storage silo. These silos are provided with vent filters (typically fabric filters) to prevent the discharge of dust along with displaced air as the silo is being filled. Venting can also be provided by a duct from the silo roof back to the precipitator inlet or hoppers. In some cases, it may be necessary to install a low pressure blower in the vent duct to overcome losses which might prevent proper release of air and cause pressure buildup in the silo or dropout of fly ash in the duct.

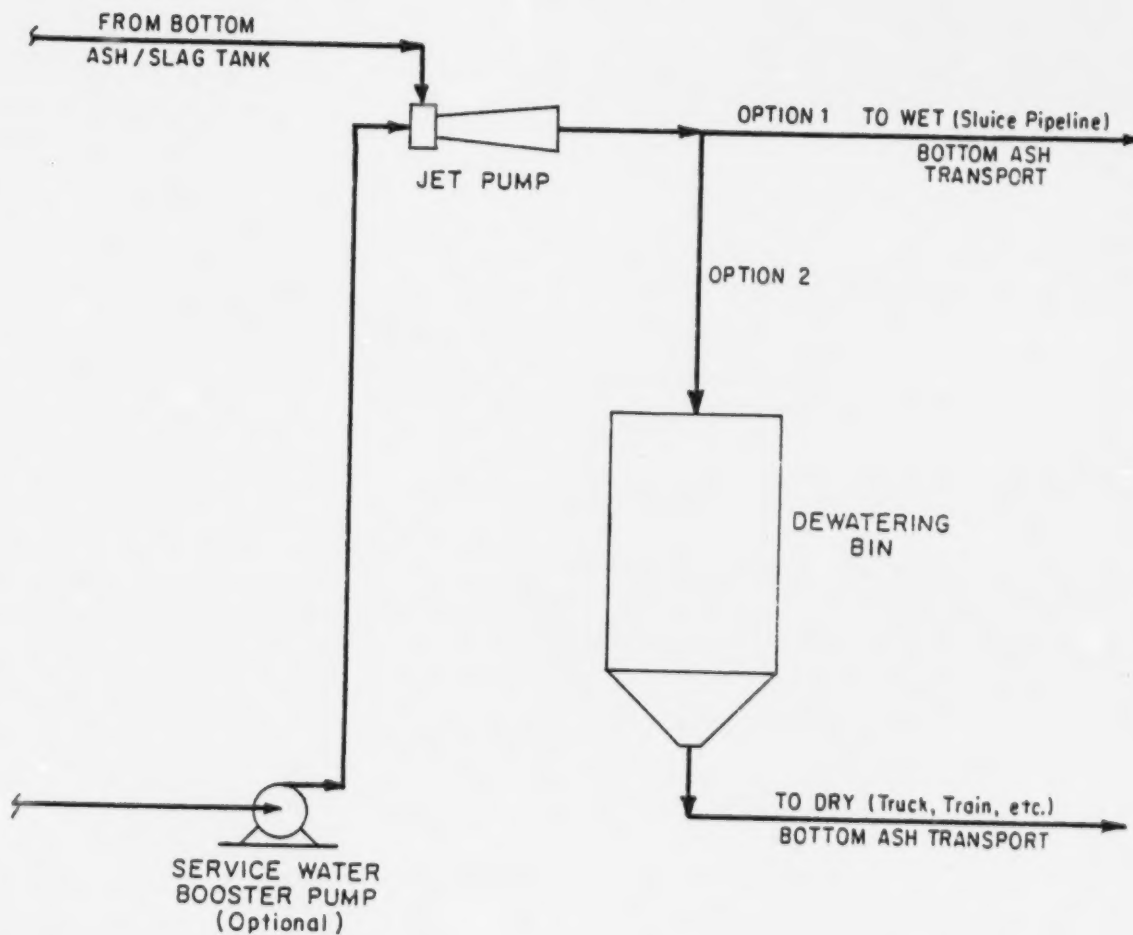
Storage silos may be of carbon steel or hollow concrete stave construction. Flat bottom silos are equipped with aeration stones or slides to fluidize dust and induce flow to the discharge outlets. Motor driven blowers supply the fluidizing air. Heaters may be required to prevent moisture from accumulating in the silo.





Source: Arthur D. Little, Inc.

FIGURE 6.13 BOTTOM ASH / SLAG HANDLING AND PROCESSING MODULE  
Wet Handling Options



Source : Arthur D. Little, Inc.

FIGURE 6.14 BOTTOM ASH / SLAG HANDLING AND PROCESSING MODULE  
Wet Handling Options

Before truck or rail transport, fly ash must be wetted to prevent dust problems during transport. This is accomplished by conditioners, which may be of the horizontal rotary pugmill type or of the vertical type. The horizontal conditioner is capable of conditioning a maximum of 45 kg/sec (180 tons/hr) of fly ash with up to 20 weight percent water addition. The vertical type can process as much as 63 kg/sec (250 tons/hr) at similar water addition rates. Figures 6.15 and 6.16 illustrate two typical fly ash storage systems.

Storage systems for dry FGD wastes will be similar to those used for fly ash.

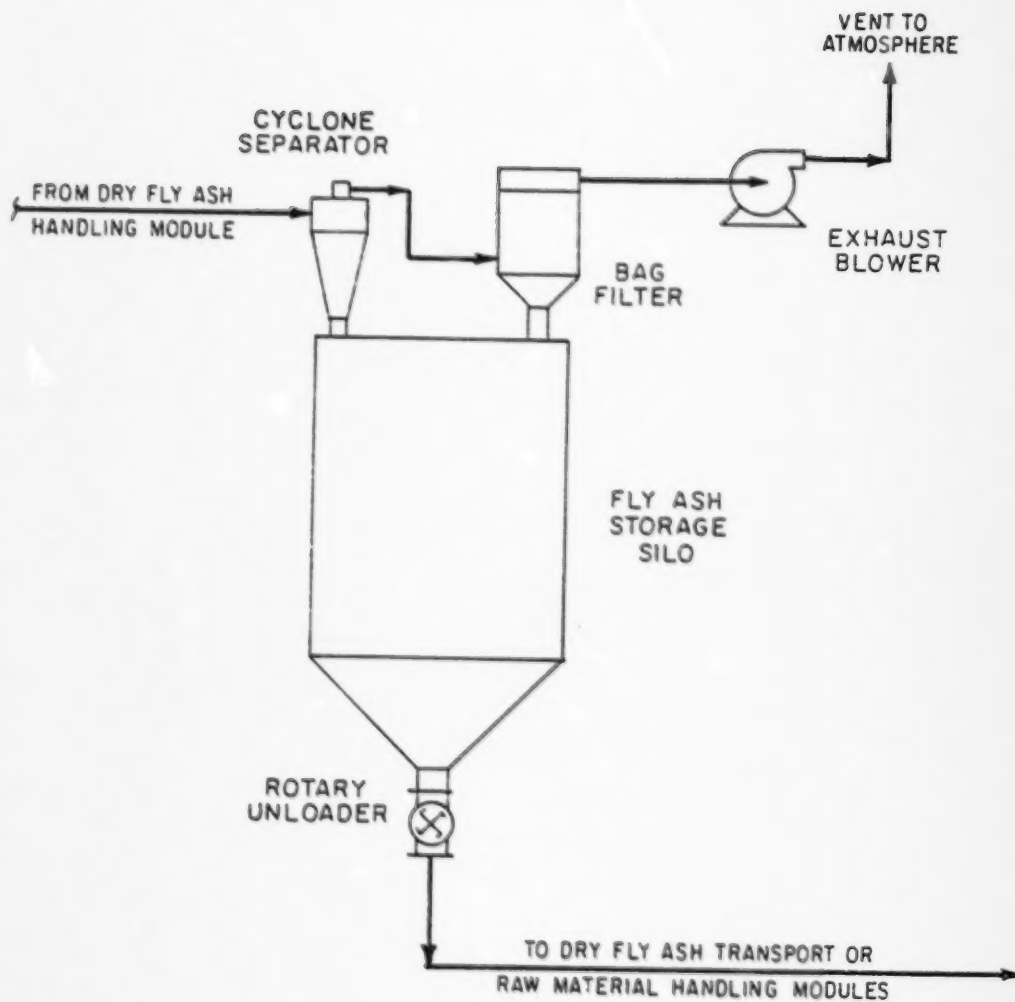
#### 6.1.5 Raw Materials Handling and Storage

In cases where FGD waste processing is achieved by stabilization or chemical fixation, cementitious additives and/or fly ash are typically added to the FGD waste. Fly ash would be considered to be a raw material in stabilization (i.e., blending) processes and in the CSI chemical fixation process. Cementitious additives include lime, which is used in the CSI and Dravo chemical fixation processes, and Calcilox®, a proprietary fixation additive in the Dravo process. Handling and storage facilities are required for all these raw materials.

These materials are commonly stored in storage bins or silos. The decision to install one or more large handling/storage units versus several small ones will be site-specific and should be based on such factors as daily additive requirements, type of demand (steady or intermittent), type of delivery, future needs, etc. In any case, the total storage capacity for purchased additives is generally at least 50 percent greater than the minimum delivery in order to guarantee adequate supply. Since lime and many fly ashes are not corrosive, storage bins or silos can in many cases be constructed from conventional steel or concrete. Lime storage units, however, must be water and air tight, since lime can absorb moisture and carbon dioxide. At present, most raw materials storage units are silos with conical bottoms. This bottom type assists in discharging the material from the silo.

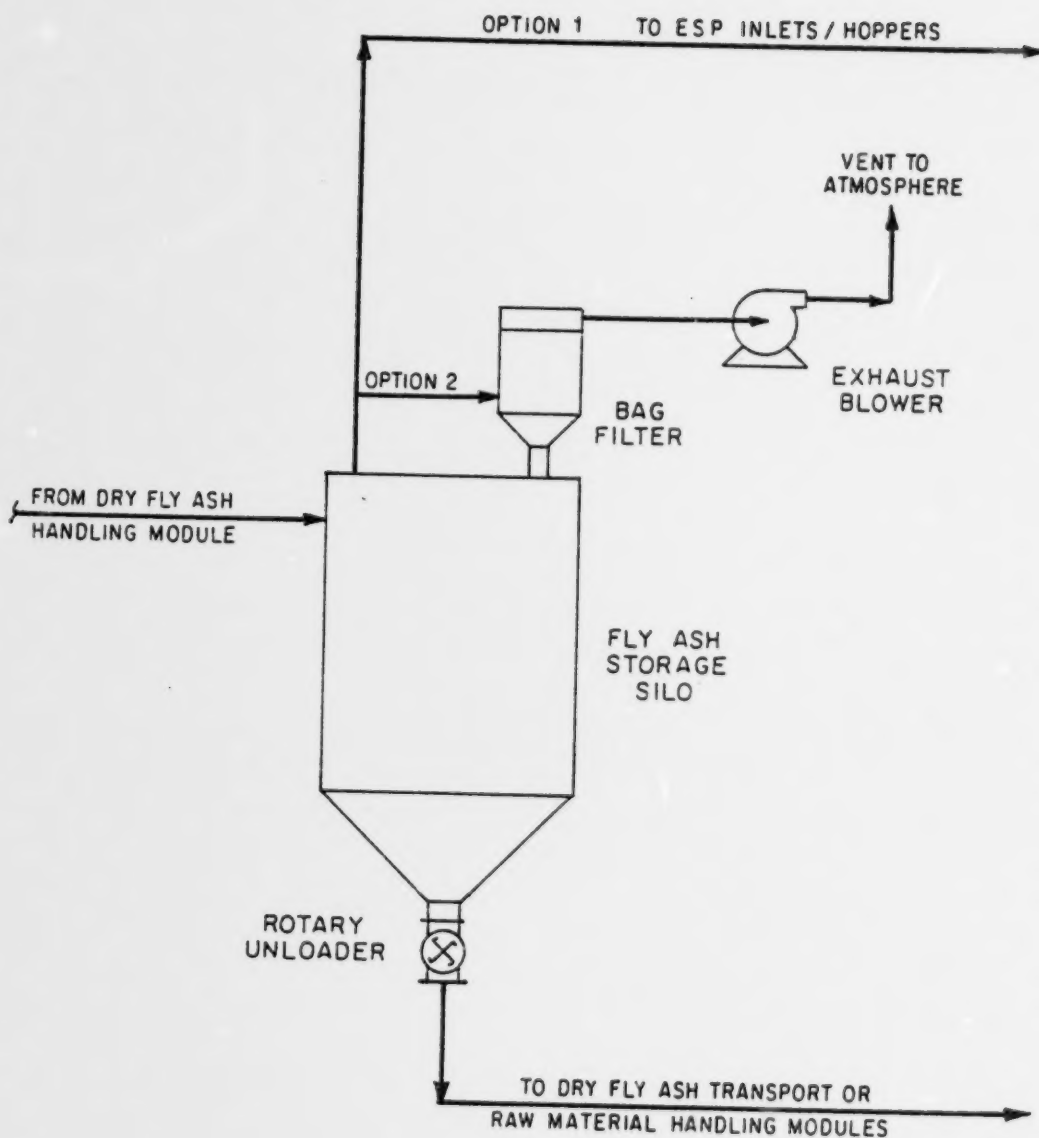
Raw material handling systems vary according to the source of the material. Fly ash is delivered to the raw material storage area from the fly ash handling or storage equipment at the power plant by pneumatic transport. The silo arrangement is like that described in Section 6.1.5 for conventional fly ash storage.

Lime and Calcilox additives are typically delivered to the raw materials handling unit in truck or rail cars. The additives can be pneumatically transported from the delivery vehicle with pressure or vacuum systems. In pressure systems, motive force is supplied by a positive displacement rotary blower mounted on the delivery vehicle. A particulate collection device must be provided to clean air vented from the silo. With the vacuum handling system, the raw material is pulled into the silo by vacuum. A lobe-type suction pump, filter receiver, and discharge air lock to



Source: Arthur D. Little, Inc.

FIGURE 6.15 FLY ASH STORAGE MODULE



Source: Arthur D. Little, Inc.

FIGURE 6.16 FLY ASH STORAGE MODULE



feed the silo must be provided. The receiver can consist of a cyclone-type collector with dust filter above, which feeds the air lock discharge. The air lock may be either a rotary driven gate or a gate lock type of feeder.

Mechanical systems are also available for handling lime and Calcilox®. With mechanical handling, the additive is usually discharged from the transport vehicle into a hopper, then transferred by a screw, pan, or drag conveyor to a bucket elevator and finally elevated to storage. Pan and drag conveyors are preferred for larger-sized additives.

Several designs are available to facilitate unloading of raw materials from storage silos. The raw materials of interest exhibit some inherent problems with flowability, necessitating the use of flow-enhancing unloading systems. The simplest device for improving flowability is the electro-magnetic vibrator attached to the outside of the hopper face. Air jets or pulsating air pads are also used on hoppers to facilitate flow. When using air jets or pads, the silo must be vented. Proper sloping of the silo bottom is another means of improving flowability.

Feeders are typically located beneath the silos, with the raw material simply flowing by gravity to the feeder. The feeding system is usually adjustable to give different rates of feed. The feed may be by volume or, more commonly, by weight. Gravimetric feeders take numerous forms but are generally classified as pivot belts, rigid belts and loss-in-weight hoppers. Most feeders for use in power plants will be of the rigid belt type. Figure 6.17 illustrates a typical additive storage and handling system.

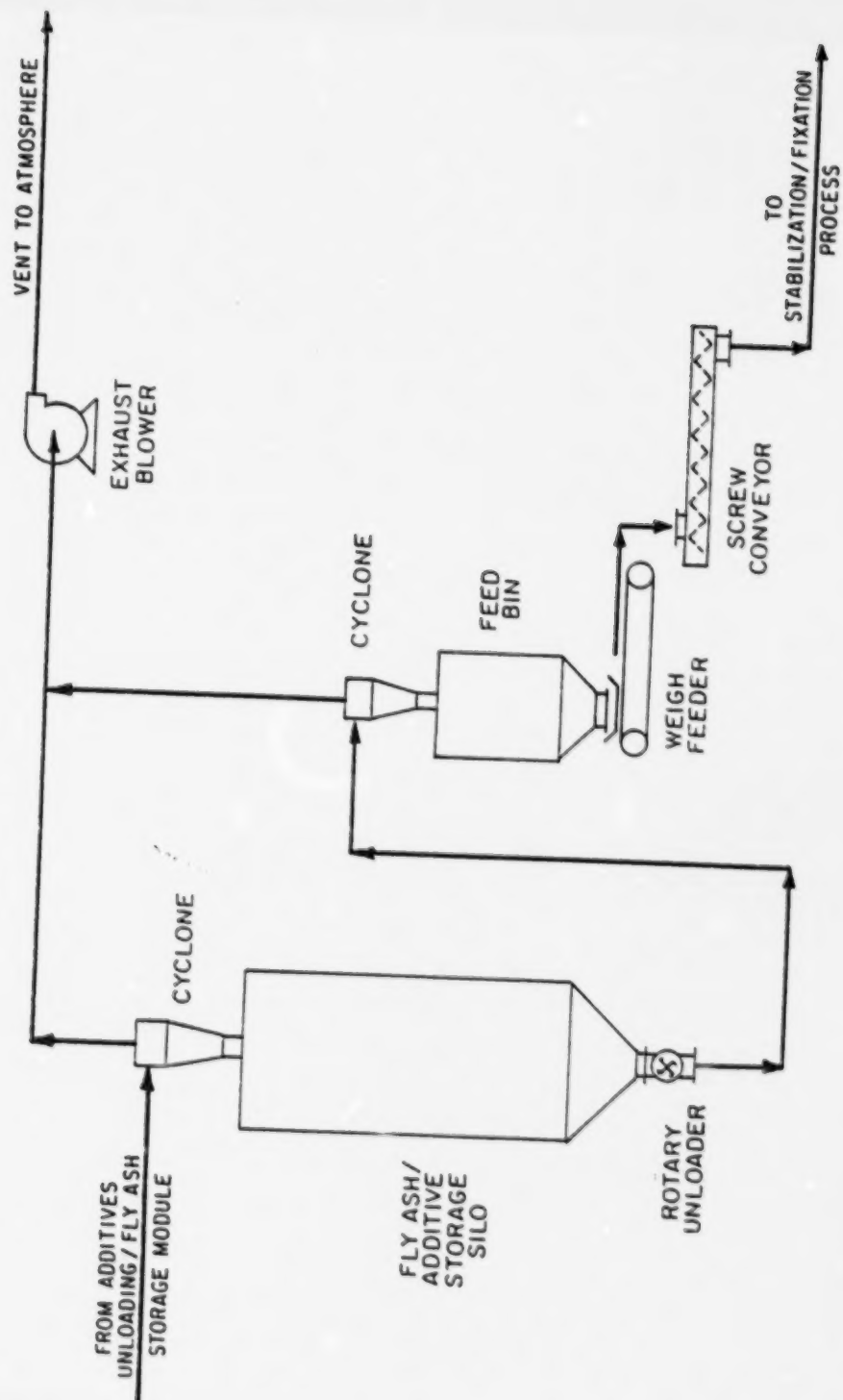
#### 6.1.6 Coal-Fired Utility Solid Waste Transport

##### 6.1.6.1 Overview--

The coal-fired utility solid waste transport system is an essential component of the waste handling and disposal operation. Conceptually, five transportation methods can be used for coal-fired utility wastes:

- belt conveyors;
- trucks;
- railroads;
- barges; and
- pipelines.

The ultimate choice of a transportation scheme depends on many factors, including waste type, distance of transport, cost, and availability of transport options. However, the form of the waste (i.e., wet or dry) ultimately limits transport options. Table 6.2 summarizes waste types and appropriate transport options.



Source: Arthur D. Little, Inc.

FIGURE 6.17 RAW MATERIALS HANDLING AND STORAGE MODULE

TABLE 6.2

## MATRIX OF WASTE TYPES AND TRANSPORT OPTIONS

| Waste Type                                    | Dry Transport Options |       |       |          | Wet Transport Option |
|-----------------------------------------------|-----------------------|-------|-------|----------|----------------------|
|                                               | Belt Conveyor         | Truck | Barge | Railroad |                      |
| Coal Ash                                      |                       |       |       |          |                      |
| • Dry                                         | P                     | ✓     | P     | ✓        | -                    |
| • Aqueous Slurry                              | -                     | -     | -     | -        | ✓                    |
| FGD Waste                                     |                       |       |       |          |                      |
| • Dry                                         | P                     | ✓     | P     | ✓        | -                    |
| • Aqueous Slurry                              | -                     | -     | -     | -        | ✓                    |
| • Primary Dewatered Waste From Wet Scrubber   | -                     | -     | -     | -        | ✓                    |
| • Secondary Dewatered Waste From Wet Scrubber | P                     | ✓     | P     | ✓        | -                    |
| • Stabilized/Fixated FGD Waste                | P                     | ✓     | P     | ✓        | -                    |

✓ = Transport method currently in use on waste type or comparable waste.

P = Potential transport option.

Source: Arthur D. Little, Inc.

#### 6.1.6.2 Transport Systems for Dry Wastes--

Dry disposal is practiced for coal-fired utility wastes which do not exhibit fluid properties. All but one (i.e., pipelines) of the utility solid waste transportation methods previously identified can be used to transport dry wastes.

Conveyors are suitable for dry waste transport but are used only for short transportation distances. To date, no belt conveyor has been used to carry coal-fired utility solid wastes from a plant to the final disposal site. This is because long belt conveyors (of several hundred meters or more) have high maintenance costs and are inflexible, permanent fixtures requiring fixed starting and ending points. A high capital investment is required for conveyor systems, although they have long service lives. For the given life of many plants, this trade-off is uneconomical.

Trucks are by far the most common and flexible means of transporting dry utility wastes over both short and relatively long distances. The availability of trucks from private contractors in varied numbers for varying time periods provides a number of solid waste transportation alternatives. All truck hauling may be contracted, or the use of a captive fleet of trucks may be supplemented by contracting additional trucks as the need arises. Three types of trucks are commonly used to transport dry FGC wastes:

- conventional rear-dump;
- semi-trailer rear-dump; and
- semi-trailer bottom-dump.

In colder areas, many trucks may be equipped with exhaust gas-heated bodies to prevent moist wastes from freezing.

The primary advantage of truck transportation for utility wastes is its flexibility. This transport method can readily accommodate changes in the quantity of solid waste produced. Additionally, transport routes may be conveniently altered in response to relocation of the disposal site. Maintenance and standby vehicles provide inherent transportation system availability; idle vehicles may be used for other hauling purposes. Contract hauling is cost-accountable as an operating expense and requires little or no capital investment.

An important disadvantage of truck transport is that this operation is labor-intensive. Thus, operating costs (due to inflation) can be expected to increase over the life of the disposal operation. Additionally, this operation has high public visibility which may result in public opposition to increased truck traffic, dust, and noise.

Transport of dry, dewatered, or processed waste is usually accomplished by open, on-highway rear-dump trucks with capacities of 7 to 18 metric tons (8 to 20 tons) or by off-highway trucks with typical capacities of 32 to 77 metric tons (35 to 85 tons). The selection of the appropriate vehicle depends on the required capacity and location of the disposal area. In cases

of transport from interim ponds to landfills which are of close proximity, self-loading scrapers may be used for both excavation and transport.

Rail transport of utility wastes is an option for long or short transport distances. It is possible to use empty coal cars to haul FGC wastes to mines for disposal, thus avoiding dead-heading costs. Today rail transport is not commonly utilized, except at a few utilities.

A major consideration with trains and trucks is the need to load the vehicles. Both vehicle types can usually be loaded with front-end loaders, belt-conveyors, or chutes. The front-end loader loads the vehicle from a surge pile by dumping directly into the car. Chutes and belt-conveyors can load directly from the outlet of processing equipment or from storage silos. However, the use of front-end loaders is preferred, since processing is usually a continuous process, whereas transportation is an intermittent process normally carried out on a 144,000 sec (40 hour) week basis.

#### 6.1.6.3 Transport Systems for Wet Wastes--

Wet disposal is practiced for coal ash and FGD wastes which exhibit fluid properties. It is theoretically possible to use all methods of transport previously described, except belt-conveyors, to transport wet wastes. However, pipeline transport of these wastes is the most practical method and, to date, the only method which has been used.

The four principal factors that must be considered in the design of pipeline systems are:

- solids settling;
- erosion/abrasion potential;
- corrosion potential; and
- freeze protection.

Of these, avoidance of solids settling in pipelines is usually the overriding factor in design. To prevent settling, conveying velocities usually range between 1.5 and 3.7 m/sec (5-12 ft/sec), depending on material density, particle size and pipeline configuration. In long pipelines handling coarse materials, velocities must be increased above those used for shorter lines. In addition, some device to create turbulence may be required to maintain a homogeneous slurry mix, particularly when conveying bottom ash or mill rejects. Fly ash and FGD waste slurries with finer particles can be pumped at the lower range of velocities.

Typically, a single pumping station is required to transport the ash and/or FGD waste slurry to the disposal pond. More pumping stations may be required for long transport distances and/or uphill transverse. At least two full-sized pipelines are required for redundancy. The need for such redundancy may be eliminated by providing emergency storage for the slurry in the event of pipeline outages. If supernatant is to be recycled to the



plant, a return pipeline (part of the waste handling and processing system) must also be provided. The waste transport pipeline may be used for this purpose if wastes are not transported through the line continuously. Provision must also be made for periodic pipeline flushing and cleaning.

Conventional steel piping and pumping equipment may be suitable for transporting utility wastes. However, when the slurry pH deviates from neutral or the slurry is highly abrasive, corrosion and erosion may be problems. In these cases, rubber-lined steel or plastic piping may be used. Steel pipe is easy to install, easy to support due to the pipe's rigidity, and has good wear resistance. Only when the slurry is highly abrasive or corrosive should one consider the use of alternatives to steel. In severely cold climates, line freezing may also be a problem that might very well decrease the potential for pipeline transport or necessitate the use of some type of steam tracing.

Ash/water slurries can be transported by centrifugal pumps or jet pumps. Slurries of about 20 weight percent ash can be moved. Considerable dilution is required at the suction end of the pump to enable slurry pumping. Centrifugal pumps are approximately 40 percent more efficient than jet pumps. This does not account for the efficiency of the auxiliary pumping equipment that supplies the ejector nozzle. However, use of jet pumps is advantageous because of the relative ease of servicing such pumps, although they must be serviced more frequently than centrifugal pumps. When system heads exceed 30.5 m (100 ft), jet pumps are impractical. Although jet pumps are not effective when operated in series, centrifugal pumps are commonly placed in series for high-head requirements.

Pumps for ash/water slurry transport are constructed of hard metals, especially in areas where abrasion is most severe. Since abrasive wear increases as velocities increase, velocities are kept sufficiently low to reduce such wear without impairing pump efficiency.

#### 6.1.7 Coal-Fired Utility Waste Placement and Disposal

##### 6.1.7.1 Overview--

Three modes of utility solid waste disposal are currently practiced:

- ponding of coal ash, FGD wastes and/or fixated FGD wastes;
- impoundment of dry or dewatered coal ash, FGD wastes and/or stabilized/fixated wastes in landfills; and
- mine disposal of coal ash and/or FGD wastes.

This discussion of placement and disposal practices is not specific to different waste types, but rather focuses on disposal practices as they are dictated by the physical form (i.e., aqueous waste slurries or dry wastes) of the waste. Table 6.3 summarizes the applicability of the various waste disposal options for specific wastes.



TABLE 6.3

## MATRIX OF WASTE TYPES AND DISPOSAL OPTIONS

|                                         | Disposal Option      |                          |                      |                       |                    |
|-----------------------------------------|----------------------|--------------------------|----------------------|-----------------------|--------------------|
|                                         | <u>Pond Disposal</u> | <u>Landfill Disposal</u> | <u>Mine Disposal</u> | <u>Ocean Disposal</u> | <u>Utilization</u> |
| <u>Coal Ash</u>                         |                      |                          |                      |                       |                    |
| • Dry Fly Ash                           | -                    | ✓                        | ✓                    | P                     | ✓                  |
| • Dewatered Fly Ash                     | -                    | ✓                        | ✓                    | P                     | ✓                  |
| • Fly Ash Slurries                      | ✓                    | -                        | -                    | -                     | -                  |
| • Dewatered Bottom Ash                  | -                    | ✓                        | ✓                    | P                     | ✓                  |
| • Bottom Ash Slurries                   | ✓                    | -                        | -                    | -                     | -                  |
| <u>FGD Waste</u>                        |                      |                          |                      |                       |                    |
| • Dry FGD Waste                         | -                    | ✓                        | ✓                    | P                     | ✓                  |
| • Stabilized/Fixated FGD Waste          | ✓                    | ✓                        | ✓                    | P                     | ✓                  |
| • Primary/Secondary Dewatered FGD Waste | -                    | ✓                        | ✓                    | P                     | P                  |
| • Primary Dewatered FGD Waste           | ✓                    | -                        | -                    | P                     | -                  |
| • FGD Waste Slurries                    | ✓                    | -                        | -                    | -                     | -                  |

✓ = Disposal option in current use for waste type of comparable waste.

P = Potential disposal option.

Source: Arthur D. Little, Inc.

Proper disposal site selection is often the most effective way to reduce the environmental consequences associated with land disposal of FGC wastes. There is no question that proper site selection can, by itself, eliminate most potential disposal impacts. A disposal site located near existing or potential groundwater users may require mitigative measures. But the same design scheme at a site which is relatively isolated from existing or potential groundwater users could be environmentally acceptable.

#### 6.1.7.2 Pond Disposal of Coal-Fired Utility Solid Wastes--

At present, most coal-fired utility plants dispose of coal ash and/or FGD wastes by ponding. However, pond disposal is anticipated to be less widely practiced in the future. This is primarily due to the relatively high cost of this disposal option. Excavation and construction of ponds is quite expensive. Thus, future ponding operations will likely be limited to sites which can be converted to ponds with relatively little excavation or dike construction.

A number of pond configurations are used for disposal of coal-fired utility wastes. The most common wet disposal configuration is the diked pond. Here the pond is contained within a perimeter embankment or dike. Figures 6.18 and 6.19 illustrate two types of diked ponds. A level or nearly level site is required for these two pond types. Dikes are constructed of materials excavated from below the existing grade at the pond site, or they may be constructed above-grade from borrow material. Above-grade construction is necessary if excavation below the existing grade is difficult due to the shallow depth of groundwater or bedrock.

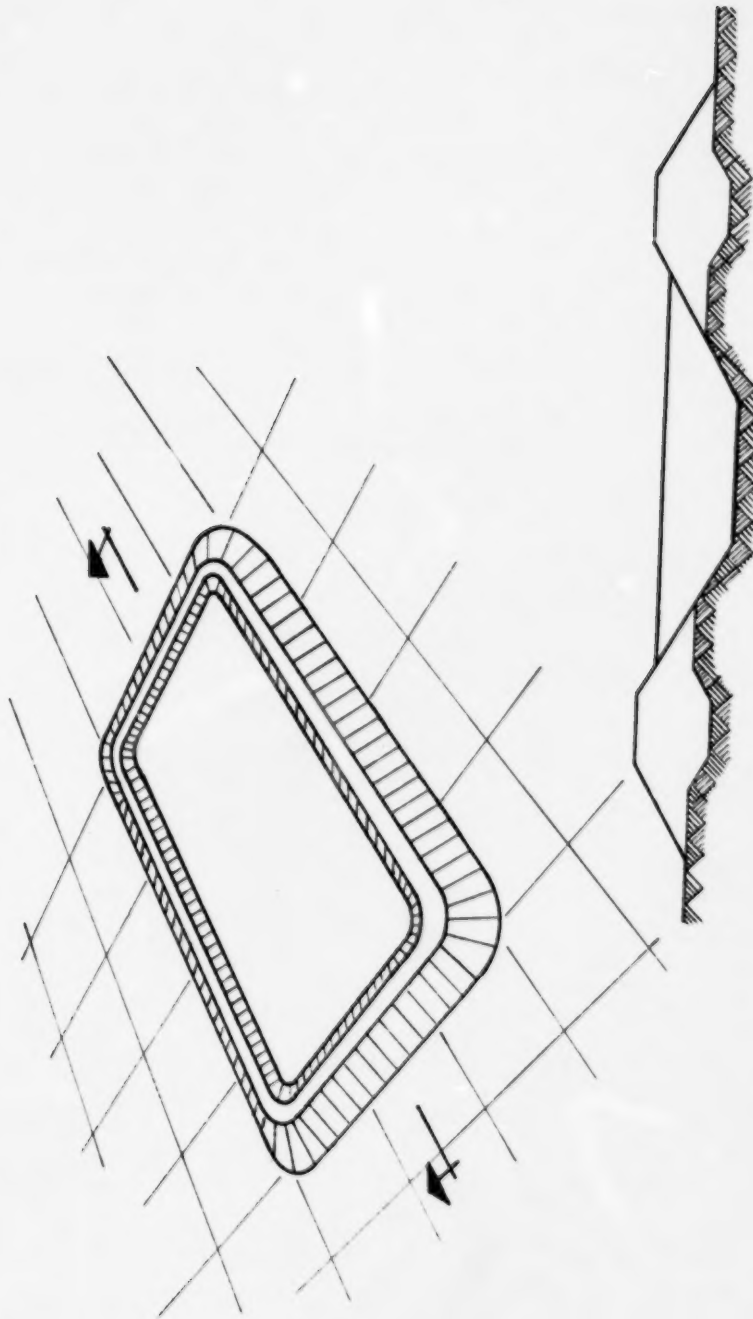
Incised ponds are contained within an excavation entirely below the existing ground surface area. This pond configuration is used where there is limited space for dike construction and where groundwater and bedrock will not be encountered in excavation. This type of pond is also attractive when excavated materials are either unsuitable for dike construction or valuable for other purposes. An incised pond design is shown in Figure 6.20.

A third pond configuration is the side-hill pond. This pond design lends itself to hilly terrain where a level site is not available for construction of incised or diked ponds. In this case, the local terrain provides one or two sides of the pond. The side-hill pond design is shown in Figure 6.21.

Other pond configurations are also in use. Existing basins, such as abandoned surface mines and quarries, may be used for pond disposal sites. Cross-valley ponds, created by constructing a dam across an existing natural valley between the two valley walls, have been employed.

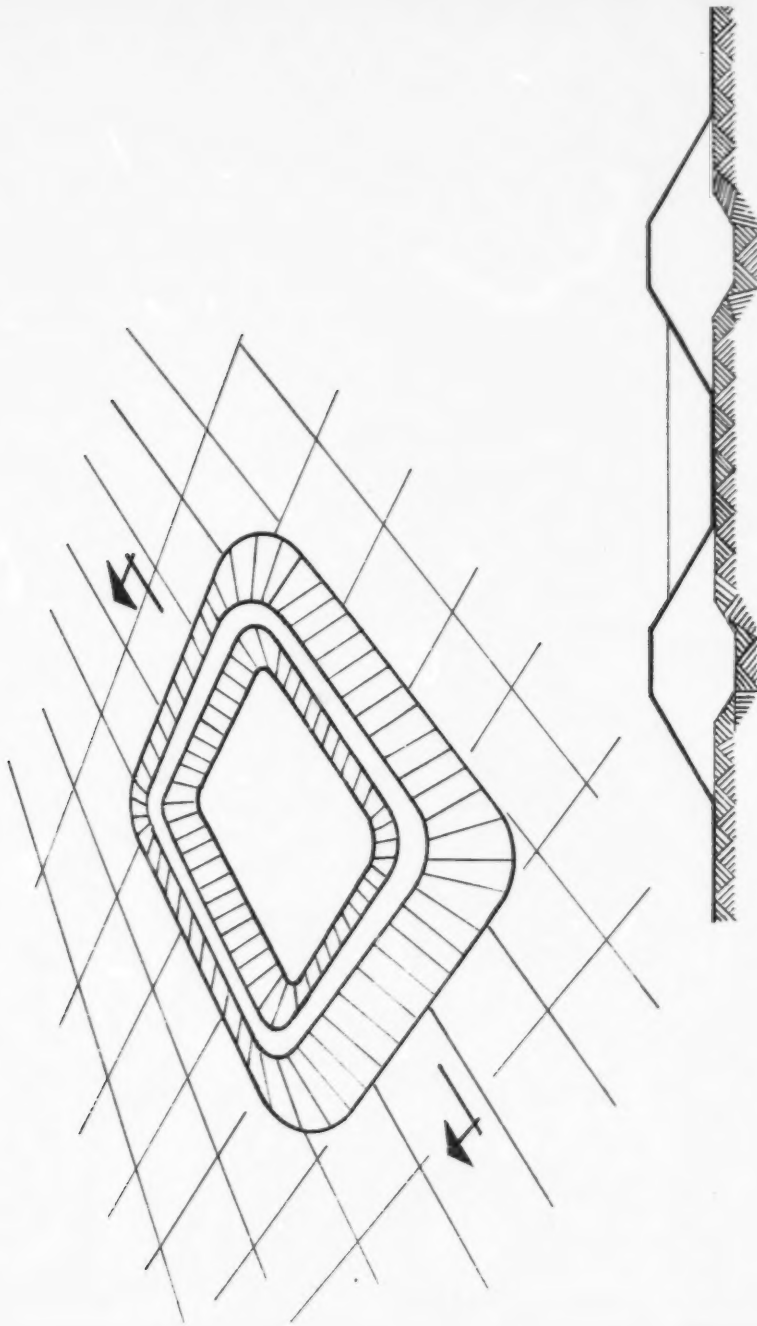
Dike stability is a critical design consideration. Designs are generally conservative and incorporate many factors which enhance stability:

- flat dike slopes;
- staged, controlled rate of construction;



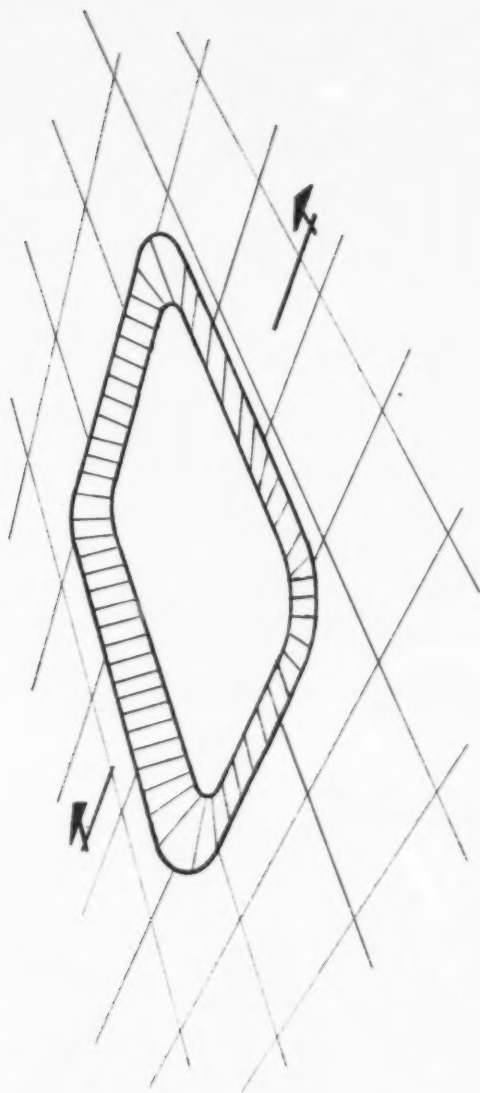
Source: Reference 17

FIGURE 6.18 DIKED POND PARTIALLY EXCAVATED BELOW-GRADE.



Source: Reference 17

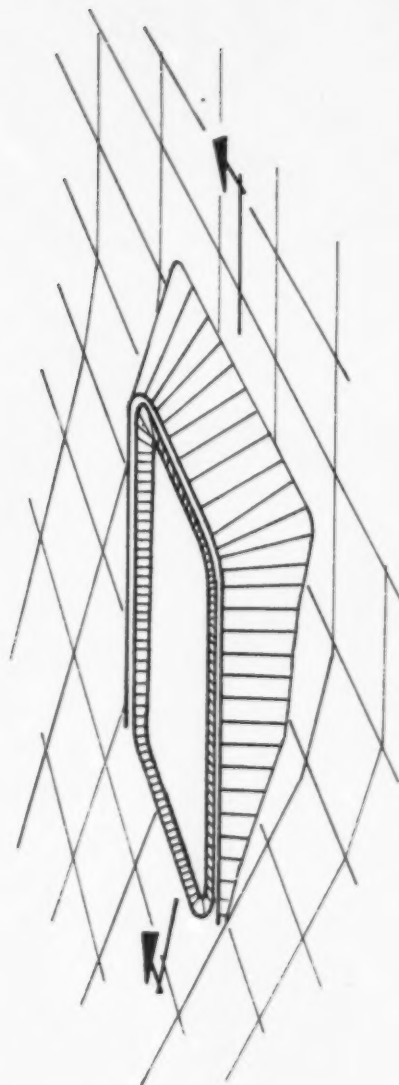
FIGURE 6.19 DIKED POND CONSTRUCTED ABOVE-GRADE.



Source: Reference 17

FIGURE 6.20 AN INCISED DISPOSAL POND.





Source: Reference 17

FIGURE 6.21 A SIDE-HILL DISPOSAL POND.

- use of high quality materials of construction;
- selection of proper material zoning to ensure self-healing in the event of cracking;
- reduction in the natural drainage area to limit the volume of run-off entering the pond to ensure balance between inflow and outflow; and
- maintenance of and design provisions for a large freeboard such that the worst conceivable inflow conditions could be contained within the pond without overflowing.

Another important consideration in pond design is the control of leachate movement. The conventional approach to this problem is to site ponds in areas where the underlying soil has low permeability. Dams are constructed of borrow material with low permeability to minimize seepage. If the permeability of the wastes and the underlying soils are not sufficiently low to control seepage to desired levels, artificial liners or sealants can be employed. Where needed, the potential options for liners are:

- Cohesive borrow, if available near the disposal site, which can be placed and compacted to a suitable depth and offer a cost-effective and efficient liner.
- Clay sealants, such as bentonite mixed with granular soils, which provide acceptably low permeabilities and can be expected to self-seal in the event of a rupture. Their main drawback, however, is a high transport cost when not available locally.
- Chemical sealants can be sprayed into the soil to plug the interstices. These can be difficult to apply, but are less expensive than clay sealants in areas where swelling clays are unavailable. However, chemical sealant liners are more permeable than those of clay.
- Synthetic liners, which are the most effective liners. They are easy to install, although punctures can seriously nullify their effectiveness. Field experience with such liners in large ponds is limited.
- Stabilized FGD wastes themselves can serve as liners.

One important point with respect to constructing either of the two earth fill-type liners (cohesive borrow and/or clay sealants) is that, presuming the liner material can be compacted to achieve a coefficient of permeability of about  $10^{-7}$  to  $10^{-8}$  cm/sec, a minimum thickness of 0.5 to 0.6 m (18 to 24 in) should be specified. This is because even if a comprehensive quality control program is enacted, it is virtually impossible to ensure that a uniform liner thickness of less than 0.5 to 0.6 m (18 to 24 in) has been placed with current construction and compaction techniques.

No reliable data are available on the long-term effects of FGC waste leachates on disposal site liner materials. The permeability of synthetic liners is claimed to be zero, but the durability of such liners in a utility waste disposal site has not been demonstrated. Clay liners have more strength, ductility, and durability than synthetic liner materials because of their inherent plasticity and buffering capacity; however, clays possess finite permeability.

As mentioned previously, wastes are transported to ponds as slurries via pipelines. The placement of wastes in ponds entails dumping the materials into the pond at a set discharge point. This type of placement can cause an uneven buildup of waste. To overcome this problem, movable tremie raft systems are sometimes used to allow for movement of the discharge line, thereby promoting uniform buildup of the wastes on the pond floor. These systems also reduce turbidity in the reservoir water. In other instances, the discharge point for the pipeline entering the pond will physically be moved from time to time over the course of the pond's life.

Bottom ash can be segregated from fly ash and/or FGD waste and used for a variety of disposal site construction purposes. A potential use for bottom ash is in constructing drains located both below and within disposal ponds. For purposes of seepage control and stability, many substantial dikes require underdrains beneath the downstream toe of the facility to control the phreatic level within the dike. In cases where a new waste basin is being built to replace an existing facility, bottom ash might be used as borrow for the underdrain. Gradations of both the bottom ash and the dike borrow will be required to verify that the bottom ash meets acceptable filter criteria. If not, a graded sand filter or filter fabric must be placed between the bottom ash and the dike.

Another potential use for bottom ash is in the construction of dewatering drains located within ponds. Most ponds are equipped with a pipe and riser outlet system which skims the top of the pond and discharges water from the basin. Often it is difficult to control the suspended solids of the water being discharged (i.e., suspended fly ash, cenospheres, etc.). A drain can be installed on the upstream face of the dike. This drain should consist of a perforated (slotted) pipe surrounded by bottom ash. The perforated pipe connects to a solid pipe which then extends through the dike. As the waste accumulates in the basin, the lower portions of the bottom ash drain may clog with the fine waste, but the water from the pond seeps into a higher level of the bottom ash drain, into the perforated pipe and out of the basin. After abandonment, such a drain will also aid in dewatering the waste deposit. In this type of application, gradations of the bottom ash will be required to determine the size of the openings (slots) in the perforated pipe.

Most state dam safety regulations require that ash basins be equipped with the means to drain water from the basins in the event of an emergency (i.e., dike instability). Decant pipes are often installed in the lowest portion of the basin and are equipped with a riser and gates to meet this requirement. These discharge pipes are usually placed lower than the plant, making recycling of the pond water impractical. The previously described

pipeline connected to the bottom ash upstream drain might have enough head to enable the pond water to be recycled by gravity back to the plant.

Deep dynamic compaction is a way to increase the density of wastes in drained disposal ponds. Utility solid wastes disposed in ponds typically do not achieve the density of similar wastes placed in landfills. Consequently, disposal of utility wastes in ponds requires a larger area than does landfill disposal of comparable quantities of waste. In addition, ponded utility wastes typically have a higher coefficient of permeability than do similar landfilled wastes. After retirement, post disposal land use of ponds is typically limited because of the soft consistency of the waste in the ponds.

Deep dynamic compaction consists of repeatedly dropping a heavy block, usually constructed of concrete, from a crane onto the area being densified. Deposits of cohesionless soil have been densified to depths as great as 9 to 12 m (30 to 40 ft) by this method. Additional disposal life can be obtained by deep dynamic compaction of previously abandoned ponds. Similarly, waste deposits adjacent to existing dikes might be densified to allow the dike to be raised by the upstream construction method and create additional disposal capacity. Finally, subsurface conditions at abandoned ponds situated in otherwise desirable areas might be improved with deep dynamic compaction to make the site suitable for development.

#### 6.1.7.3 Landfill Disposal of Coal-Fired Utility Solid Wastes--

Dry FGC waste disposal methods can involve any one of the following operations:

- dry collection and direct landfilling of coal ash and dry sorbent FGD wastes;
- dewatering and landfilling of coal ash and FGC wastes; and
- landfilling of fixated and stabilized FGC wastes.

The exact nature of a particular operation depends on the type of waste disposed and disposal site.

Most dry disposal operations involve landfilling. Landfill design may be based on three common configurations:

- The heaped landfill, which is structurally the simplest form of fill. This type of landfill is typically used in areas with level terrain. Although this design offers advantages in terms of slope stability, minimal site preparation requirements, and minimized groundwater pollution, it does not blend with the surrounding terrain and is, therefore, aesthetically undesirable.
- The side hill landfill design, which is often used in areas of hilly terrain where the natural slope of one side of a hill or valley may provide containment. The side hill landfill must be prepared properly to ensure stability. Properly constructed, these landfills blend well with many existing terrains and can provide valuable property when reclaimed.



- The valley fill design, which is the most common type of landfill. It is often the most complex in terms of original site preparation. Natural valleys or ravines are often sources of surface water runoff and may have springs along side slopes. In such cases, surface water and groundwater control is usually necessary to avoid water accumulation and a potential leachate problem. Drainage must be provided and, in some cases, hydrologic modifications are needed to divert water flow around the landfill.

The three common landfill configurations are shown in Figures 6.22 through 6.24.

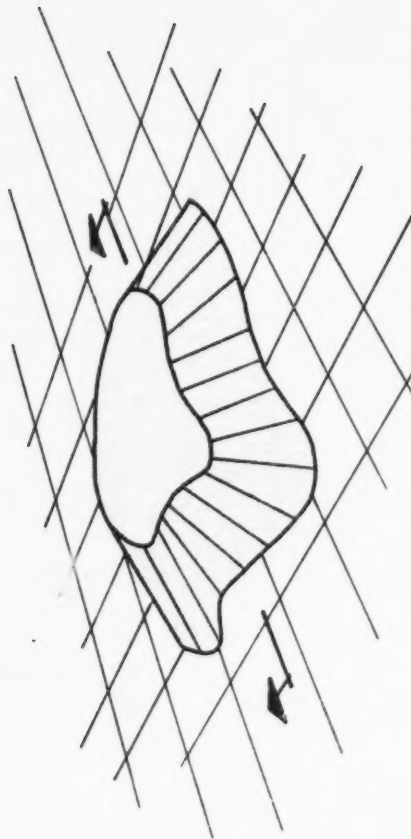
Proper landfill design requires control of both leachate movement and runoff. The former may be controlled through the use of liners such as those discussed in Section 6.1.7.2, although this is not presently a common practice. Stabilization and fixation processes reduce waste permeability, thereby reducing leachate movement. However, to achieve full environmental benefits from waste processing and compaction during placement, proper design is necessary to control runoff. Conventional runoff control practice consists of retaining runoff in temporary retention basins, thereby allowing suspended solids to settle prior to discharge or use of the overflow.

Because landfilled wastes are disposed of in essentially dry form, the physical stability measures required for disposal of wet wastes (i.e., dike construction, etc.) are not applicable. Unlike pond disposal, the focus of landfill design, then, is not on construction. Instead, operations that ensure the inherent stability of the placed wastes take precedence. In general, such operations at the landfill area involve dumping, spreading and compaction. Usually, only small sections of the landfill are worked at any one time.

Spreading equipment can include dozers, loaders, scrapers, and/or graders. Use of the crawler dozer is usually preferred, due to its firm grip regardless of weather conditions. In applying dewatered scrubber sludge to a landfill, the sludge is brought to the site and spread into thin layers which are usually less than 0.7 m (2 ft) thick. Dry coal ash is generally spread in lifts ranging from 0.3 to 1 m (1 to 3 ft). The spreading is normally confined to as small an area as possible to maximize equipment utilization and control dust or mud. In most instances, two areas will be alternately used. Trucks haul the wastes to one area while dozers spread the wastes in the second area. Lifts are built up to a total ultimate fill height which is site specific but generally may range from 9 m (30 ft) to over 25 m (82 ft).

Once the waste has been spread, it may be compacted to increase its density and strength. Various types of compaction equipment are used:

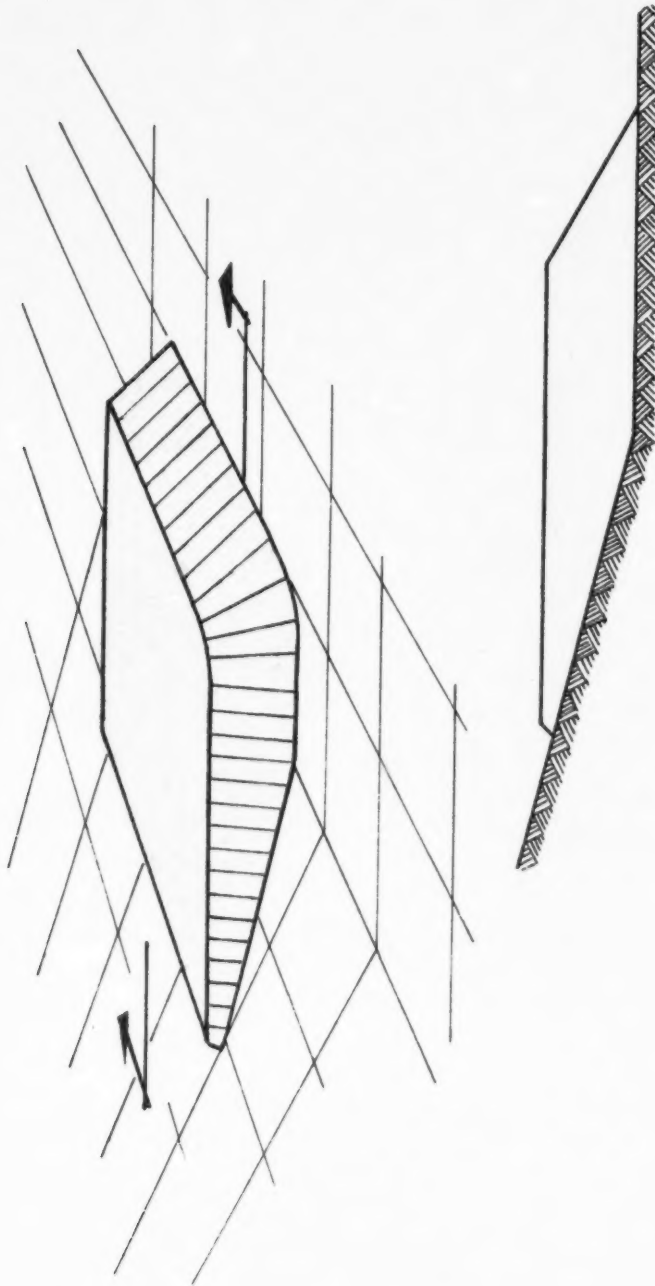
- rubber-tired rollers;
- segmented pad compactors; or
- vibratory compactors.



Source: Reference 17

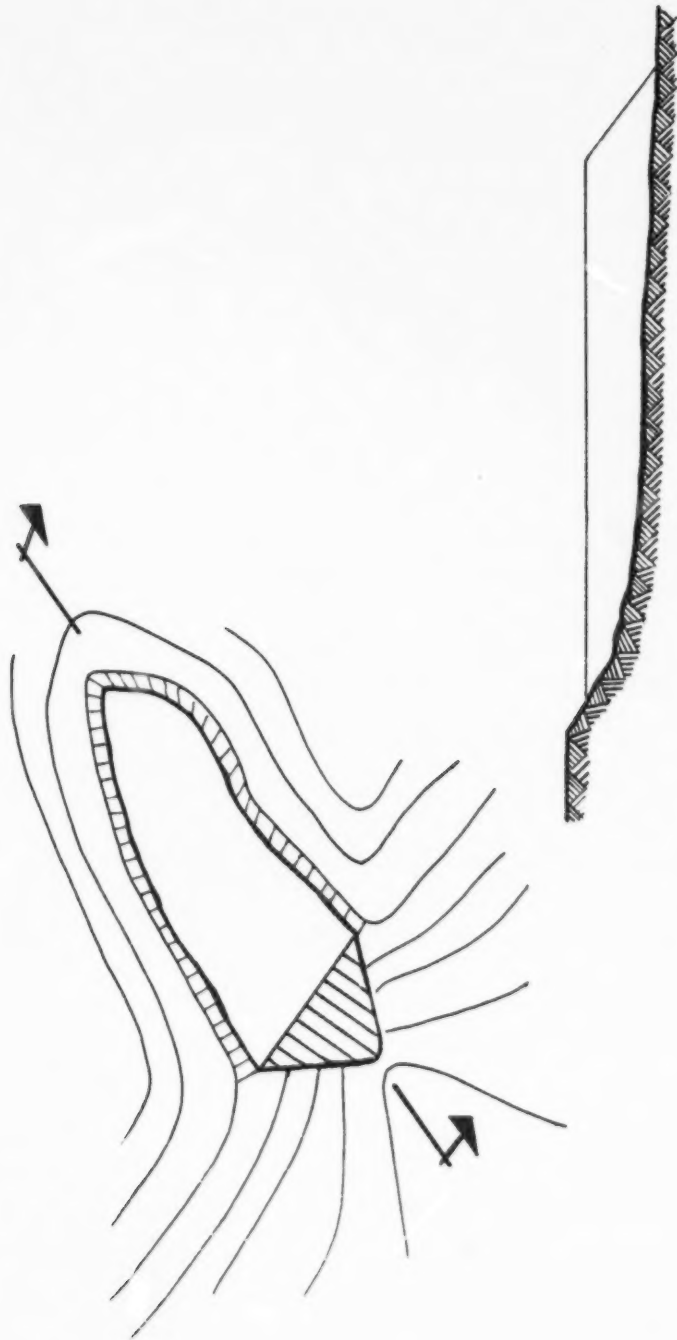
FIGURE 6.22 A HEAPED LANDFILL CONFIGURATION.





Source: Reference 17

FIGURE 6.23 A SIDE-HILL LANDFILL.



Source: Reference 17

FIGURE 6.24 A VALLEY-FILL DISPOSAL CONFIGURATION.

The type of compactor is very dependent on the type of waste being landfilled. Compactors are available in both self-propelled and towed models. However, where placement rates exceed 75 kg/sec (300 tons/hr), a self-propelled model is usually preferred.

Segregating bottom ash from the FGC waste stream and using it in selected applications controls water migration from utility waste landfills. A common problem at many FGC waste landfills is natural seepage (from springs, for instance) and surface infiltration through the fills. This water often collects in pervious zones of the landfill and slowly migrates out of the fill as leachate. Placing pervious bottom ash within the lowest levels of the FGC waste fill can be an inexpensive way to create an underdrain beneath the waste deposit. Rainfall that infiltrates through the fill and water from natural seeps can be collected by the bottom ash drain and transported to the toe of the fill. At this point, the water can be collected and treated if necessary or otherwise discharged.

Physical instability is a potential problem for all utility wastes, including stabilized wastes. Geometric factors such as height and slope angle in a waste/ash fill are interrelated. Stability depends on the combination of fill height, slope angle, and waste properties (i.e., density, degree of saturation, effective cohesion, effective angle of shearing resistance, and behavior during shearing). For a given material, safe fill height decreases with increasing slope angle.

Underlying materials may lead to instability of a waste deposit if the stresses in those materials exceed the strength of the materials under the site-specific loading conditions (i.e., failure may occur because of the weakness of the underlying strata). Weak compressible soils would be potential problem materials in this context.

Instability problems may be ameliorated by compaction, which produces several changes: elimination of voids (between chunks, not between individual particles); increase of waste density; bonding of particles moved closer together through stabilization reactions; and increase of effective stress and residual total stress levels. The stability of settled sulfite wastes under compaction equipment will vary with the solids content of the wastes (assuming no chemical stabilization of wastes) and probably with the type of compaction equipment (static, rubber-tired, steel roller, sheepfoot, vibratory, etc.). Sulfite-rich FGD wastes tested to date may be incapable of supporting compaction equipment of any kind unless high solids contents are achieved first. Liquefaction may be a more serious threat in sulfate-rich FGD wastes, especially under vibratory loading.

During dry periods, dust can be a problem. A water truck equipped with a rear spray bar should suffice for controlling road dust. In cold, dry climates, the use of calcium chloride crystals may be more effective.

#### 6.1.7.4 Mine Disposal of Coal-Fired Utility Solid Wastes--

Disposal of coal-fired utility wastes in worked-out mines is receiving increased attention. Surface coal mines and underground room and pillar

mines for coal, limestone, or lead/zinc ores offer potential, although coal mines, and in particular surface area coal mines, are the most likely candidates for waste disposal, since they offer the greatest disposal capacity and are frequently tied directly to power plants. Many new coal-fired power plants are "mine-mouth" (located within a few miles of the mine), with the mine providing dedicated coal supply. Since the quantity (volume) of coal ash and FGD wastes produced is considerably less than the amount of coal burned, such mines typically would provide disposal capacity throughout the life of the power plant.

Disposal of coal ash and FGD wastes in surface mines is a reasonably well developed technology, although the practice is not widespread. In general, inactive surface mines are considerably less promising than active mines for waste disposal. Unreclaimed surface mines can be used for disposal of wastes between remaining spoil banks which offer suitable sites for disposal. However, because of recent surface mine reclamation legislation, the number of sites and total capacity for wastes available in the future will be limited.

In active surface mines, there are three options for coal ash and FGD waste placement:

- in the working pit, following coal extraction and prior to return of overburden;
- in the spoil banks, after return of overburden but prior to reclamation; and
- mixed with or sandwiched between layers of overburden.

The overriding consideration with respect to disposal at active mines is that minimal disruption of mining or reclamation activities should occur. This provides a number of constraints on the disposal system. The physical condition (consistency) of the wastes must be amenable to handling, transport, and placement by earth-moving equipment, with minimal potential impact on the mining operations. For pit-bottom disposal this means that the wastes at the time of placement or immediately thereafter should have as a minimum the consistency of a soil-like material with little or no liquefaction potential. A slurry-like material, or a waste with a tendency to flow either when placed or when overburden is dumped on top of it, could present significant operational problems or unacceptably high costs for containment measures. More leeway exists for disposal in V-notches (between spoil banks); however, here again soil-like materials or relatively cohesive materials that are relatively easily handled and transported will result in the least cost and minimal disruption of reclamation activities. At the least, FGD wastes must be well filtered (55 percent solids or higher), admixed with dry fly ash, or treated.

The amount (volume) of waste disposed in any surface mine should not exceed the amount of coal removed. The objectives of strip mine reclamation include returning the mined terrain to topographic configurations similar to

the original terrain. Returning significantly more waste to a mine than coal extracted could slow down the mining and reclamation activities. In most cases, this does not represent a constraint, since the wastes returned to the mine will be only those resulting from burning the coal removed. Depending on the type of coal, the particulate and sulfur oxide control system, and emission standards to be met, the volume of total dry waste will vary from less than 10 percent of the coal burned to slightly over 50 percent.

Finally, minimal use should be made of existing mine equipment for transport and placement of the wastes at the mine. Dedicated equipment is preferred and, in most cases, mandatory. In almost all scenarios, waste is most easily placed by truck dumping. The use of coal trucks for this purpose could lead to unacceptable delays in coal mining operations due to the additional time for waste loading and discharging (and possibly cleaning operations). Furthermore, most large mines use large bottom-dump trucks for coal haulage. These are designed to carry as much as 135 metric tons (150 tons) of coal and are usually constructed of aluminum. Some types of FGC waste can corrode aluminum; additionally, the bottom dumping of wastes would be impractical. Such trucks are not designed for easy maneuvering, and operating them (or any other equipment) on a freshly dumped layer of waste would be difficult. The type of truck used for transport and placement will greatly affect the quantity of waste that can be easily disposed.

Pit-bottom dumping is probably the most common, simplest, and least disruptive method of surface mine disposal. Transport and placement of the wastes is most easily accomplished with rear-dump trucks. Access to the pit floors is usually good, since roads are maintained in relatively good condition for coal mining and hauling equipment. Also, there is usually adequate time for waste placement between coal removal and overburden replacement.

Major potential problems are interference with coal removal in the working pit due to instability of adjacent spoil banks caused by the underlying wastes and the possible congestion in the working pit due to the two-truck transport systems (one for coal and one for wastes). Most of these problems are avoidable by proper scheduling of mining and disposal activities and control of waste properties and placement. In some cases, it may be advisable to concentrate waste dumping on the side of the pit farthest from the highwall (against the newly created spoil bank) to provide a region where no waste exists when the next cut is taken and the overburden dumped on the waste.

V-notch or spoil bank disposal, as with pit-bottom disposal, would involve truck dumping, and many of the constraints described above for waste characteristics and disposal operations will apply. This method involves somewhat more effort than pit-bottom disposal, since roads must be cut into the spoil banks at the base of the vees to allow access by waste trucks. This can be accomplished with relative ease in most cases with standard dozers and road-grading equipment. While it requires more effort, it may also offer some advantages in that the disposal is less likely to impact directly the coal removal operation. There is less pit congestion, less



potential of creating spoil bank instability, and generally less stringent scheduling requirements. For these reasons, spoil bank disposal may often be preferred over pit-bottom disposal.

Waste disposal operations can also be adapted to allow mixing of the waste and overburden. This can be accomplished readily in contour strip mines where haulback methods are used and in some western mines where overburden is handled by truck. However, in many strip mines, mixing waste and overburden or sandwiching waste between layers of overburden may require too much additional handling of materials and place added constraints on the dragline or shovel operation. Thus, in most mines, this type of operation is not expected to be practical.

Many disposal system configurations are possible, depending on the type of waste, distance between the power plant and mine and the type of placement. In general, though, the distance between the mine and power plant is the overriding factor which dictates the amount of handling and the types of transfer facilities required.

Underground mine disposal is not presently well developed. Methods developed for backfilling certain materials (mine spoil or coal processing wastes) in active mines can presumably be modified for use with coal ash and FGD wastes. Hydraulic and pneumatic flushing of fly ash into inactive portions of active and abandoned mines has been practiced.

#### 6.1.7.5 Monitoring of Coal-Fired Utility Solid Waste Disposal Sites--

The monitoring of an FGC waste disposal site may consist of air, surface water, and groundwater monitoring. The typical minimum number of water monitoring points are:

- Surface water
  - ash pond discharge;
  - sedimentation pond outlet;
  - upstream surface flow; and
  - downstream surface flow.
- Groundwater
  - upgradient at one point; and
  - downgradient at three points.

Surface and groundwater monitoring programs involve collecting samples from specific locations over a set period of time. In monitoring surface water, samples are taken directly from the body of water by manual or automatic means. Groundwater monitoring is usually carried out by providing access to groundwater via monitoring wells. Such wells are installed by drilling contractors. Pumping equipment, either above ground or in the well, is required to obtain samples. An alternate method of obtaining samples is with bubble samplers. Such wells incorporate two tubes, one which is pressurized with gas during sampling, thereby causing the groundwater in the second tube to be moved to the surface.



#### 6.1.7.6 Reclamation of Coal-Fired Utility Solid Waste Disposal Sites--

It is difficult to specify the ultimate use of land on which utility waste disposal is planned or practiced. However, it is important to recognize engineering constraints on post-closure land use. Post-closure land use tends to be limited by nature of the loads created or the sensitive nature of the structures or facilities to be built on the site. For example, placement of fill in a uniform layer over the entire waste deposit is likely to be feasible, but imposition of concentrated loads (e.g., footings in a building) might cause bearing failure with rapid plunging of the loaded element into the wastes. Vibratory loads as from machinery foundations could have adverse effects on unfixated FGD wastes. The rate of application of load also is extremely important--loads applied slowly may allow consolidation of the wastes during loading, yielding higher strength than for material loaded rapidly. Rapid application of load increases shearing stresses and may decrease shearing resistance if the structure of the wastes is disturbed.

Reclamation of coal ash and FGD waste disposal sites generally consists of covering the disposal site with suitable soil and/or clay materials followed by revegetation. Erosion control measures may also be taken.

Reclamation plans for coal ash and FGD waste disposal sites should be carefully formulated and account for a number of factors, including:

- federal and local effluent guidelines for sites generating point-source discharges (these standards are applicable even upon retirement of a site);
- proposed site uses following closure of the disposal site;
- surrounding land use which may affect the viability of the proposed site uses following closure;
- climatological effects which influence the selection of plant species for revegetation, planting schedules and erosion control structure;
- local vegetation which may provide insights into the types of plants suitable for use in revegetation;
- surrounding soil types which could be employed for cover material;
- water erosion, both due to raindrops and surface runoff, and the ability of proposed vegetation cover to curtail such effects; and
- slope lengths and gradients which may induce or reduce erosion.

The proposed post-closure uses for a disposal site will, to some extent, determine the requirement of the reclamation program. Not all sites are suitable for all possible post-closure uses, since the stability of the placement is critical to some end uses.

Three categories of soils can be used for disposal site reclamation. These include: 1) topsoil, 2) friable subsoil, and 3) low permeability cover lining materials. Of primary concern in reclamation is determination of the required soil cover depth. In some cases, state regulations mandate a minimum cover depth of 0.6 m (2 ft). At sites where such regulations are not in effect, less soil cover can be used. However, the quality of soil cover material will influence the depth required to support vegetation and, therefore, will influence the quantity required for reclamation. The cost of reclamation is highly dependent upon the quantity of soil cover required.

Typically, three layers of material are applied to the graded disposal site in the following sequence:

- an impermeable layer to serve as a protective seal preventing recharge and leachate generation (if mandated by state/local regulation or deemed necessary by the utility);
- an intermediate subsoil with good rooting characteristics; and
- an upper topsoil zone with high biological activity and nutrient content.

Most blade-type machinery may be used to spread soil cover, although care must be taken not to compact the upper two layers of soil to a density which results in poor root permeability or low soil oxygen content. The lowest soil layer, which, if required, should be impervious, should be densified as much as possible to minimize permeability. These layers should air dry following spreading but before compaction to prevent crust formation which makes plant growth difficult.

Plants for revegetation should be well adapted to site conditions; native species would, in most cases, be suitable. In cases where seeds for such plants are unavailable or prohibitively expensive, species which can inexpensively provide adequate surface coverage and which are adaptable to site soil and climate conditions can be used. Such species must be readily available and capable of propagating. Finally, plants employed for revegetation must be compatible with post-closure land use.

Following final soil cover placement, fertilizer, lime, or other soil additives may be applied to the site. Disk harrowing of the surface to a depth of 0.25 to 0.20 m (6 to 8 in) can be used to mix such additives. Seeds are planted in a uniform distribution by one of four common methods:

- broadcast,
- seed drill,
- hydroseeding, and
- aerial seeding.

Of these methods, hydroseeding is most commonly used. In this method, seed and, in some cases, fertilizer and mulch are sprayed on the surface as a slurry. Broadcast and aerial seeding are undesirable methods of seed application, since distribution is generally non-uniform. Seed drill application cannot be used on many sloping areas, but is a good method in terms of distribution and depth of the seed placement. Following seeding, mulching is commonly required to protect the area from soil erosion and to provide a good environment for germination and plant development. Hay, straw, wood chips, and other artificial materials may be used as mulching materials. With hay or straw, stabilization methods to prevent loss of the mulching materials are required. These methods include chemical binding of these materials together and crimping them to the surface to secure their position.

Runoff flow entering the reclaimed site should be limited. In some instances, a system of diversion ditches and waterways has been constructed to prevent erosion from runoff. Additionally, entrapment structures which limit the amount of sediment leaving a site, such as settling ponds, check dams, vegetative buffers, and filters, may be used.

## 6.2 COSTS OF COAL-FIRED UTILITY SOLID WASTE HANDLING AND DISPOSAL

### 6.2.1 Overview

A variety of studies have been undertaken to analyze the costs associated with handling and disposing solid wastes from coal-fired power generation. Generalized studies of FGC process technology have recently incorporated analyses of associated waste handling and disposal costs and a number of conceptual design and cost studies of these systems have been undertaken by government agencies and private organizations, notably EPA and EPRI. Additionally, evaluations of site- or system-specific solid waste handling and disposal operations have been performed by a number of utilities, engineering firms and other consulting organizations.

Unfortunately, the design and operating assumptions in these economic studies, as well as the battery limits for the disposal systems evaluated in the various studies, differ considerably. Costs are reported for a wide range of conditions: different plant sizes; different coal types; various operating load factors and plant lives; different degrees of waste processing; and different unit cost factors and base years. In addition, cost estimates in the literature often represent hypothetical plants for which the engineering designs and cost premises underlying the cost estimates are unknown. Also, costs are generally presented in lump-sum form covering an entire waste handling and disposal facility, thereby eliminating any real use of that data when trying to estimate the cost of a generic waste handling and disposal system.

Hence, direct comparisons and use of these cost estimates for projections are difficult, at best. Consequently, until now, no one data base existed that enabled one to estimate easily, quickly and accurately the

capital and operating costs of coal ash and/or FGD waste handling and disposal systems. This section provides a generic cost estimation methodology which can serve as the basis for developing these costs.

## 6.2.2 Capital and Operating Cost Curve Development

### 6.2.2.1 Background--

In recognition of the need for a quick, easy, and accurate method for preparing conceptual capital and operating cost estimates for the waste handling and disposal system of a proposed new coal-fired utility plant, a set of cost curves was developed as part of this project. The use of conceptual cost estimating curves is especially desirable when more detailed cost estimates are not required. The set of cost curves presented here is structured in such a way that one can estimate both the capital and annual modular costs for the handling, processing, storage, transport, and ultimate disposal of any or all of the three major wastes (i.e., fly ash, bottom ash, FGD waste) generated at coal-fired power plants. In this way, for a given combination of utility plant size (i.e., electric power generating capacity) and waste type(s), one can utilize the cost curves to estimate (late-1982 basis) the capital and operating costs for the entire waste handling and disposal system or for any of the individual corresponding process modules comprising that system.

The cost curves presented here were developed with both the engineering cost data base developed for the six disposal sites evaluated during this project (see Section 5) and engineering cost data found in the literature.

In evaluating the site-specific engineering/cost data for utilization in the development of the generic waste type process module capital and annual cost curves, it became obvious that this cost data would require adjustment. This would put the cost data developed for these six sites and the cost data found in the literature on a common basis. In other words, to the extent practicable, the cost data were adjusted to eliminate any effect caused by unusual site specific conditions. For example, in developing the generic capital cost curve for the transport of ash by sluice pipeline it was necessary to ensure that all data points (both from the open literature and our six site evaluation) were all on the same engineering design basis [i.e., a plant to disposal pond distance of 1.6 km (1.0 mile)]. This required adjusting the length of piping, number of valves, etc. and ultimately the capital cost associated with these elements for a number of case study examples.

The type of plant and/or disposal site-specific factors and conditions which had to be evaluated and adjusted to a common basis ranged from transport distances to site preparation activities. Some of the more important site-specific factors which required noting and/or adjustment before the corresponding cost data developed from the six site engineering/cost assessment could be used are summarized below:



- More complex piping was required at the Allen Plant for coal ash handling due to presence of five small units (totaling 1155 MW) in contrast to newer plants which would probably employ only two larger units (600 MW) to achieve the same generating capacity.
- The pressure (pneumatic) conveying system used at the Dave Johnston Plant utilizes atypical, expensive Nuva Feeders to feed fly ash from electrostatic precipitator hoppers into pressurized conveying lines.
- Coal ash haul trucks in use at the Dave Johnston Plant are off-the-road type which are more costly to purchase.
- Site preparation at the Dave Johnston Plant included excavating more soil than would be required for use in reclamation activities, as was not the case at other plants with landfill operations.
- Distance to transport coal ash and FGD wastes at the Elrama Plant (from plant to landfill) is 19.2 km (12.0 miles), while the common basis for most studies is 1.6 km (1.0 mile).
- No clearing or grubbing was required at the Elrama landfill, since this area was the site of an abandoned strip mine.
- FGD waste handling/processing at the Elrama Plant included maximum dewatering using thickeners, vacuum filters and pug mills.
- FGD waste handling/processing at the Sherburne County Plant included less dewatering using thickeners.
- Distance to transport coal ash and FGD wastes at the Sherburne County Plant (from plant to ponds) is 0.4 km (0.25 mile), while the common basis used is 1.6 km (1.0 mile).
- Coal ash and FGD waste disposal ponds at the Sherburne County site are clay-lined.
- Coal ash landfill (gravel pits) at the Powerton site are lined with Poz-O-Pac®.
- Bottom ash handling/processing at the Powerton Plant includes dewatering bins.
- Coal ash and FGD handling system at the Sherburne County site includes a supernatant recycle system.
- The coal ash handling system at the Smith plant includes a supernatant recycle system.
- Distance to transport coal ash at the Smith Plant (from plant to pond) is 0.4 km (0.25 mile), while the common basis used is 1.6 km (1.0 mile).



- The fly ash to bottom ash ratio for the Powerton Plant is 40/60, because cyclone-fired boilers are used. Pulverized coal-fired boilers generally result in a 80/20 fly ash to bottom ash split.

All these factors were evaluated and often resulted in adjustments being made, when practicable, to various process module capital and annual costs. Following the adjustments, both the unadjusted and adjusted data points were used to develop and then evaluate the generic curves and their band.

Waste disposal cost data in the literature were compiled, evaluated, and adjusted, to the extent practicable, and updated to late-1982 dollars (to put them on the same basis as data developed for the six study sites). Table 6.4 lists the sources of cost data that were utilized. While the cost data found in the literature were adjusted to put all data on a common basis, the data were obtained from many different studies, all of which used many different assumptions. Taken collectively, however, the various cost data can be and were used to determine the approximate, overall range of costs expected for the handling and disposal of utility wastes. This was accomplished by plotting the appropriate cost data and using a combination of engineering judgement and the mathematical method of least squares curve fitting to obtain the most appropriate curve with the best fit for the given data. These curves are presented as bands which incorporate the variations in costs due to system variations. In some cases, due to a paucity of cost data, it was not possible to define the exact limits of the cost curve bands. In these cases a conservative approach was taken in which variations were compared to those of other systems and a conservative factor applied to the curve fit by the least squares method.

This simplified method for determining utility waste disposal system costs is intended to provide conceptual cost estimates which have accuracy limits of plus 30 percent and minus 10 percent; these should never be substituted for more detailed estimates prepared from a site-specific engineering cost data base. At the level of accuracy for which conceptual estimates are intended to be used, effects due to such site-specific factors as geology, soil conditions, terrain, land availability, local labor rates, etc., are not specifically addressed. Consequently, it should be recognized that this simplified cost estimating procedure is a tool for planning purposes where the effects of such site-specific conditions have a less significant impact on the desired cost accuracy.

The basic engineering design and economic premises for which these curves are applicable are summarized in Tables 6.5 and 6.6, respectively. From Table 6.5, it can be seen that the cost curves presented are applicable for estimating the capital and operating cost (within the accuracy limits noted above) for the waste handling and disposal of fly ash, bottom ash and/or FGD waste from new coal-fired utility plants with generating capacities ranging from 200 to 3,000 MW. The boiler is assumed to be of the pulverized coal-fired (dry bottom) type which produces a fly ash/bottom ash ratio of 80/20. The ash and sulfur contents of coal employed range from 12 to 15 percent and 0.5 to 3.5 percent, respectively. Particulate removal is accomplished by ESPs, and sulfur oxides removal is attained by conventional

TABLE 6.4

SUMMARY LISTING OF FGD WASTE DISPOSAL ENGINEERING COST STUDIES UTILIZED IN DEVELOPING COST CURVES

| Contractor                           | Sponsor | Waste Types Evaluated |           |                      | Disposal Options Evaluated |          |      | Reference No. |
|--------------------------------------|---------|-----------------------|-----------|----------------------|----------------------------|----------|------|---------------|
|                                      |         | Coal Ash              | FGD Waste | Coal Ash & FGD Waste | Pond                       | Landfill | Mine |               |
| Arthur D. Little, Inc.               | EPA     | ✓                     | -         | ✓                    | ✓                          | ✓        | -    | -             |
| Arthur D. Little, Inc.               | EPA     | -                     | -         | ✓                    | -                          | -        | ✓    | 5             |
| Aerospace Corp.                      | EPA     | -                     | -         | ✓                    | ✓                          | ✓        | -    | 49            |
| Aerospace Corp.                      | EPA     | -                     | -         | ✓                    | ✓                          | ✓        | -    | 50            |
| Aerospace Corp.                      | EPA     | -                     | -         | ✓                    | ✓                          | ✓        | -    | 16            |
| Environmental Research & Tech., Inc. | EPRI    | ✓                     | -         | ✓                    | -                          | -        | ✓    | 51            |
| Engineering Science                  | DOE     | ✓                     | -         | ✓                    | ✓                          | ✓        | ✓    | 52            |
| GAI Consultants, Inc.                | EPRI    | ✓                     | -         | -                    | ✓                          | ✓        | -    | 21            |
| Michael Baker Jr., Inc.              | EPRI    | ✓                     | -         | ✓                    | ✓                          | ✓        | ✓    | 53            |
| Michael Baker Jr., Inc.              | EPRI    | -                     | -         | ✓                    | ✓                          | ✓        | -    | 17            |
| Michael Baker Jr., Inc.              | EPRI    | -                     | -         | ✓                    | ✓                          | ✓        | -    | 54            |
| Tennessee Valley Authority           | EPA     | ✓                     | -         | -                    | ✓                          | ✓        | -    | 55            |
| Tennessee Valley Authority           | EPA     | -                     | -         | ✓                    | ✓                          | -        | ✓    | 56            |
| Tennessee Valley Authority           | EPA     | -                     | -         | ✓                    | -                          | ✓        | -    | 57            |
| Tennessee Valley Authority           | EPA     | -                     | -         | ✓                    | ✓                          | ✓        | -    | 58            |

Source: Arthur D. Little, Inc.

TABLE 6.5

SUMMARY OF BASIC ENGINEERING DESIGN PREMISES FOR GENERIC COST CURVES  
FGC WASTE HANDLING AND DISPOSALENGINEERING DESIGN PREMISESPower Plant

|                                           |                                  |
|-------------------------------------------|----------------------------------|
| New or Retrofit                           | New                              |
| Plant Size (MW) <sup>a</sup>              | 200-3000                         |
| Boiler Type                               | Pulverized Coal-Fired Dry Bottom |
| Heat Rate (M joules/kWh; Btu/kWh)         | 10-12; 9,750-11,400              |
| Location                                  | United States                    |
| Service Life (yr)                         | 30                               |
| Load Factor (Lifetime Average Percentage) | 70                               |
| Fly Ash/Bottom Ash Ratio                  | 80/20                            |

Coal Properties

|                                     |                         |
|-------------------------------------|-------------------------|
| Sulfur Content (Percent)            | 0.5-3.5                 |
| Ash Content (Percent)               | 12.0-15.0               |
| Heating Value (M joules/kg; Btu/lb) | 22.0-26.7; 9,500-12,000 |

Air Pollution Control

|                                   |                                                                |
|-----------------------------------|----------------------------------------------------------------|
| Particulate Control               | ESPs                                                           |
| Particulate Removal (Percent)     | >99                                                            |
| Sulfur Oxides Control             | Conventional Lime Scrubber with<br>or without Forced Oxidation |
| Alkali Stoichiometry              | 1.1                                                            |
| SO <sub>2</sub> Removal (Percent) | >70                                                            |

Disposal Site

|                                       |                                                  |
|---------------------------------------|--------------------------------------------------|
| Type                                  | Pond/Landfill                                    |
| Design Life (yr)                      | 30                                               |
| Terrain                               | Level                                            |
| Groundwater Monitoring Wells (Number) | 6                                                |
| Reclamation (Closure)                 | 0.45 m cover soil; 0.15 m top<br>soil; reseeding |
| Liner                                 | None                                             |
| Distance from Plant (km; miles)       | 1.6; 1.0                                         |

<sup>a</sup> Large plants employing multiple 500 MW units.

Source: Arthur D. Little, Inc. estimates.

TABLE 6.6

SUMMARY OF BASIC ECONOMIC PREMISES FOR GENERIC COST CURVES  
FGC WASTE HANDLING AND DISPOSAL

Economic Premises

|                                                  |                                                   |
|--------------------------------------------------|---------------------------------------------------|
| Year of Capital and Annual Cost Estimates        | Late-1982                                         |
| Capitalization of Site<br>Construction (Percent) | 100                                               |
| Capitalization of Site<br>Reclamation (Percent)  | 100                                               |
| Capital Charge Factors <sup>a</sup>              |                                                   |
| • Process Equipment and Disposal Site            | 0.147                                             |
| • Mobile Equipment                               | 0.280 <sup>b</sup>                                |
| • Reclamation                                    | 0.035                                             |
| System Battery Limits                            | Waste Handling/Processing<br>to Ultimate Disposal |
| Land Cost (\$/m <sup>2</sup> ; \$/acre)          | 0.79; 3200                                        |

<sup>a</sup>These capital charge factors were employed in the preparation of the first year annual costs for the six study sites and in the adjustments made to cost data found in the literature, when practicable.

<sup>b</sup>The capital charge factor for mobile equipment includes an allowance for interim replacement.

Source: Arthur D. Little, Inc., estimates.

lime or lime/forced oxidation scrubbing. Disposal options include both ponding and landfilling, with a distance of 1.6 km (1 mile) between the plant and the disposal site. The disposal site proper, whether a pond or landfill, will include a total of six groundwater monitoring wells. In addition, the waste placement/disposal module will include the cost for site reclamation, which will entail covering the pond(s) (naturally dewatered) and/or landfill(s) with 0.45 m (1.5 ft) of cover soil followed by 0.15 m (0.5 ft) of top soil with subsequent seeding and mulching.

The most important economic premises (in Table 6.6) to note include the following:

- Both capital and annual costs are reported in late-1982 dollars.
- Annual capital charges for process equipment and the disposal site were taken as 14.7 percent of the total capital investment.
- Annual capital charges for mobile equipment were taken as 28.0 percent of the total capital investment (wherever practicable).
- Annual capital charges for reclamation were taken as 3.5 percent of the total capital investment.
- Capital costs include 100 percent capitalization of site construction and site reclamation.
- The system battery limits begin at in-plant waste handling and go up to and include the placement and ultimate disposal of the waste.

Factors for capital charges were determined using the methods proposed by the Electric Power Research Institute (EPRI)(43). These methods are discussed below.

#### 6.2.2.2 Process Equipment/Disposal Site--

Capital charges for the coal ash and FGD waste handling and disposal system (exclusive of mobile equipment and disposal site reclamation) were assumed to be 14.7% of the total capital investment on the basis of a 30-year operating life. The capital charge factor was calculated as follows:

|                                                    |         |
|----------------------------------------------------|---------|
| • Total Return (Weighted Cost of Capital)          | 10.00%  |
| • Book Depreciation (Sinking Fund)                 | 0.61%   |
| • Allowance for Retirement Dispersion              | 0.56%   |
| • Levelized Annual Income Tax                      | 4.70%   |
| • Levelized Annual Accelerated Depreciation Factor | (2.47)% |
| • Levelized Annual Investment Tax Credit           | (0.77)% |



• Property Taxes, Insurance, etc.

2.00%

LEVELIZED ANNUAL FIXED CHARGE RATE\*

14.63%→14.7%

#### 6.2.2.3 Mobile Equipment--

Mobile equipment capital charges were calculated to be 28%, based on a 10-year mobile equipment life. This took into account the need for interim replacement of mobile equipment within the 30-year operating life of the plant.

The EPRI Technical Assessment Guide discusses interim replacements (pp. VII-2, VII-3 and VII-19, Reference 43). The approach outlined in the EPRI guide and that used in this study result in identical capital charges for mobile equipment, although the means of calculating these differs slightly.

EPRI defines an equivalent capital cost as:

$$EC = BPC + (F)(CR) \quad (1)$$

where:

EC = equivalent capital cost

BPC = cost of plant that is not affected by interim replacement

CR = cost of plant that requires interim replacement

F = factor that puts all plant capitalization on a common time basis

In this case capital charges are calculated as:

$$CC = (X)(EC) \quad (2)$$

where:

CC = capital charges

X = capital charge factor for total plant life

Alternatively, a capital charge factor for interim replacements was defined separately from the factor for items that have lives equal to the plant life. This factor was defined as:

$$XR = (X)(F) \quad (3)$$

where:

\*Based on 30-year book life and 20-year tax life. Flow through accounting used. (See page V-9, Reference 43.)

XR = capital charge factor for items that require interim replacement

In this case, capital charges were calculated as:

$$CC = (X)(BPC) + (XR)(CR) \quad (4)$$

This equation is equivalent to equation (2) when the expression for EC is substituted:

$$\begin{aligned} CC &= (X)(EC) \\ &= (X) [BPC + (F)(CR)] \\ &= (X)(BPC) + (X)(F)(CR) \\ &= (X)(BPC) + (XR)(CR) \end{aligned} \quad (5)$$

Thus, the annual costs for mobile equipment used in this study are consistent with the EPRI Guide (43).

Further, F is defined as:

$$F = \left[ \frac{CRF(r, N)}{CRF(r, LR)} \right] \left[ \frac{FCR_R}{FCR_P} \right] \times \left\{ 1 + \frac{n_r - 1}{p - 1} \left[ \frac{(1 + i)}{(1 + r)} \right]^{p \times LR} \right\} \quad (6)$$

where:

CRF = capital recovery factor at discount rate, r for life, N = plant life or LR = replacement life

$FCR_j$  = fixed charge rate for j, j = R or P

R = interim replacement

P = plant

i = inflation rate

r = discount rate

(p.VII-9, Reference 43)

In this case:

$$F = \left( \frac{0.1061}{0.1627} \right) \left( \frac{0.19}{0.14} \right) \left\{ 1 + \left( \frac{1.06}{1.10} \right)^{10} + \left( \frac{1.06}{1.10} \right)^{20} \right\} \quad (7)$$

and

$$XR = (0.147) (1.9) = 0.28 \rightarrow 28\% \quad (8)$$

#### 6.2.2.4 Reclamation--

Capital charges for reclamation were estimated as the annual annuity payment (assuming 10% earnings on investment) required to produce a fund sufficient to pay for reclamation costs inflated over the 30-year life of the disposal site. The annual payment, estimated at 3.5% of the total capital cost for reclamation, was calculated as follows:

$$F = P (1 + i)^N \quad (9)$$

where:

F = future cost of reclamation in year, N

N = life of plant

P = present value of plant

i = inflation rate

In order to produce a fund of magnitude, F, at the end of the plant's service life an annuity paid annually is required:

$$A = F \left( \frac{r}{(1 + r)^N - 1} \right) \quad (10)$$

where:

r = rate

N = duration of payments

Equations (9) and (10) can be combined to yield:

$$A = P (1 + i)^N \left( \frac{r}{(1 + r)^N - 1} \right) \quad (11)$$

The capital charge factor, then, is the factor that when applied to the present value of reclamation yields an annual cost (i.e., an annuity) that can be invested to yield the desired fund at the end of the plant's service life. In this case the factor, Z, is:

$$Z = (1 + i)^N \left( \frac{r}{(1 + r)^N - 1} \right) \quad (12)$$

For our study:

$$Z = (1 + 0.06)^{30} \left( \frac{0.10}{(1 + 0.10)^{30} - 1} \right) \quad (13)$$

$$Z = 0.035 \rightarrow 3.5\%$$

### 6.2.3 Capital and Annual Cost Curves

Capital cost curves in late-1982 dollars, with a range of absolute accuracy of plus 30 percent and minus 10 percent, were developed. As previously noted, the cost curves presented here are based on two primary data bases: (1) the engineering cost data base developed for the six waste disposal sites evaluated during this project; and (2) engineering/cost data available in the open literature. Of these two data bases, more emphasis and a higher level of confidence was placed on the data resulting from the evaluation of the six waste disposal sites, since these estimates were based on detailed process descriptions, process flow diagrams, material balances, equipment lists, and equipment specifications. Most engineering information was developed during site visits and through contacts with plant personnel who provided engineering data, as explained in Section 4.

The total fixed capital investment includes direct capital investment for equipment, installation and service facilities; piping and insulation; transport lines; foundations and structures; site preparation and earthwork; roads; electrical and instrumentation; buildings and mobile equipment (trucks and earthmoving equipment). Materials, labor/fabrication and installation costs are also included.

The total capital investment also includes indirect capital investment. Indirect capital investment is comprised of items such as contractor's overhead, contractor's profit, engineering design and supervision, architect/engineering fee, allowance for start-up, and contingency. The total capital investment consists of the total fixed investment plus allowances for start-up modifications and the cost of land, site monitoring and site reclamation activities.

First year annual cost estimates in late-1982 dollars were developed on the basis of 6,100 hours per year of operation at full load. First year annual costs include both direct and indirect costs. Direct costs are the cost for such items as: raw materials, utilities (i.e., process water, electric power, diesel fuel and chemicals), operating expenses (e.g., operating and mobile equipment labor, supervision and maintenance materials and labor, etc.), and subcontracted items (e.g., pond dredging). Indirect costs include such items as payroll and plant overhead expenses and capital charges.

Generic capital and annual cost curves were developed for the various waste types and process modules discussed in Section 6.1. Due to the wide number of system design variations available within any given module, it was, in some cases, appropriate to provide more than one curve for certain waste type and module combinations. Even with such provisions, each curve presented represents the costs of a set of system designs. Thus, a band of costs, rather than a single cost curve applies. The remainder of this section presents the generic cost curves and discusses the engineering premises for each of the curves and the effect of process variations on costs.

Capital cost curves (capital cost versus power plant size, MW) were developed for fly ash and bottom ash handling and disposal; similar curves (capital cost versus FGD waste generation, metric tons) were developed for the handling and disposal of FGD waste. Cost curves for the handling and disposal of FGD wastes were plotted with capital cost versus FGD waste generation rate rather than power plant size. This is because FGD waste generation rates vary widely for any given plant size; therefore, the cost of FGD waste handling and disposal is too dependent on the sulfur content of coal used for any one plant size to have a meaningful cost relationship in terms of plant size. Similarly, annual cost curves (annual cost versus power plant size, MW, and annual cost versus ash generation, metric tons/year) were developed for fly ash and bottom ash waste handling and disposal; in addition, cost curves (annual cost versus FGD waste generation, metric tons/year) were developed for the handling and disposal of FGD waste. For the same reason as mentioned above, cost curves for the handling and disposal of FGD waste were not developed in terms of annual cost versus plant size, MW.

#### Fly Ash Handling and Processing

For the purpose of developing generic cost curves, we identified three process design variations for fly ash handling and processing:

- wet handling of fly ash without recycle of supernatant;
- wet handling of fly ash incorporating supernatant recycle; and
- dry fly ash handling.

It was necessary to segregate the wet and dry fly ash handling options, since the design criteria and costs of these vary significantly and a single cost band could not account for the cost differences between these two options. Additionally, the use of supernatant recycle systems in conjunction with wet fly ash handling also impacts design requirements and system costs. Thus, it was appropriate to develop costs for these two options separately.

Cost differences for fly ash handling also arise within each of the three options noted above. Considering wet fly ash handling, exclusive of recycle provisions, system design (and thereby cost) variations arise primarily as a result of differences in fly ash collection modes (i.e., wet collection by scrubbers versus dry collection by ESPs and baghouses). In many wet collection systems, fly ash and FGD waste are collected simultaneously. In such systems the water processing (i.e. dewatering) requirements are dictated primarily by the presence of FGD wastes rather than by the fly ash itself; in many cases fly ash in these systems undergoes significantly more processing than would typically be required. Costs for such processing is relatively expensive, both in terms of capital and annual expenditures. This cost assessment does not address fly ash dewatering, either by interim ponding or the employment of mechanical dewatering devices (e.g., thickeners), due to the paucity of available cost data for such



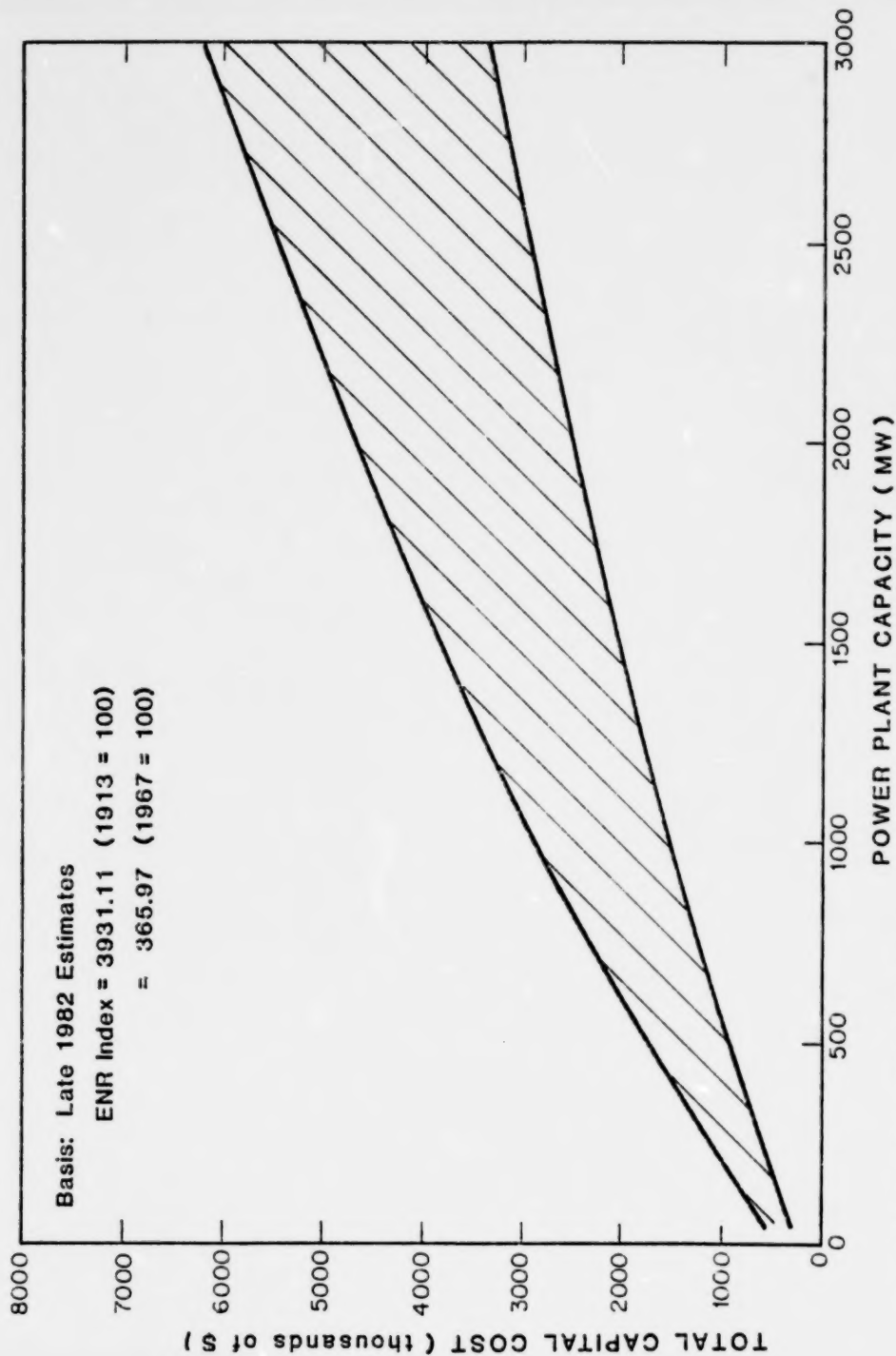
systems. It is important to note, however, that the use of interim ponds for dewatering fly ash is not widely practiced. Also, use of this practice is not expected to grow in the future. Additionally, fly ash is typically only dewatered via mechanical dewatering devices when collected simultaneously with FGD waste. At the present time, the simultaneous collection of fly ash and FGD waste (e.g., in the case of fly ash alkali scrubbing) is not a common practice and is not expected to grow significantly in the future.

Wet handling systems for fly ash collected by scrubbers, exclusive of any dewatering provisions, are relatively less costly than those for dry wastes, since such systems do not require the slurring equipment required for wet handling of dry wastes. In general, the cost of simpler wet fly ash handling systems (e.g., handling of fly ash collected in scrubbers without dewatering) would be expected to fall in the lower ranges of the cost curve band, while more complex systems (e.g., vacuum pneumatic conveying of dry fly ash to a slurring device) would exhibit costs in the higher regions of the band. Figures 6.25 (capital costs), 6.26 and 6.27 (annual costs) provide generic cost curves for wet fly ash handling where no supernatant recycle is practiced.

The discussion of costs for wet fly ash handling systems with supernatant recycle closely parallels that for similar systems which do not include recycle provisions. The only major difference between these two types of systems is the recycle system which is, itself, relatively simple in design and has few design alternatives. Thus, the discussion in the preceding paragraph details the major concerns with respect to the cost curves for fly ash handling systems which incorporate supernatant recycle; these curves are provided as Figures 6.28 (capital costs), 6.29 and 6.30 (annual costs).

Dry fly ash handling also has many process variations. The simplest and least costly systems are used for short conveying distances; vacuum systems are typically chosen for this service. For longer conveying distances, pressure pneumatic conveying is required. Such systems can be very costly due to the expense associated with equipment required to feed fly ash from the ESP or mechanical collector hopper discharge to the conveying line, as well as the costs associated with the large amount of piping required for long conveying distances. In some cases, vacuum/pressure combination systems are used to overcome the operating difficulty associated with feeding ash from the hoppers. In general, costs are related to conveying distances, with costs in the lower portion of the capital and annual cost curve bands relating to shorter conveying distance. The generic capital cost curve is presented as Figure 6.31, while annual costs are shown in Figures 6.32 and 6.33.

Bottom Ash Handling and Processing -- Two sets of capital and annual cost curves which pertain to two handling and processing options are provided for bottom ash. In one option [Figures 6.34 (capital costs), 6.35 and 6.36 (annual costs)] wet handling of bottom ash without supernatant recycle is



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.25 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY**  
**FLY ASH HANDLING and PROCESSING**  
**( Wet Handling Without Recycle )**

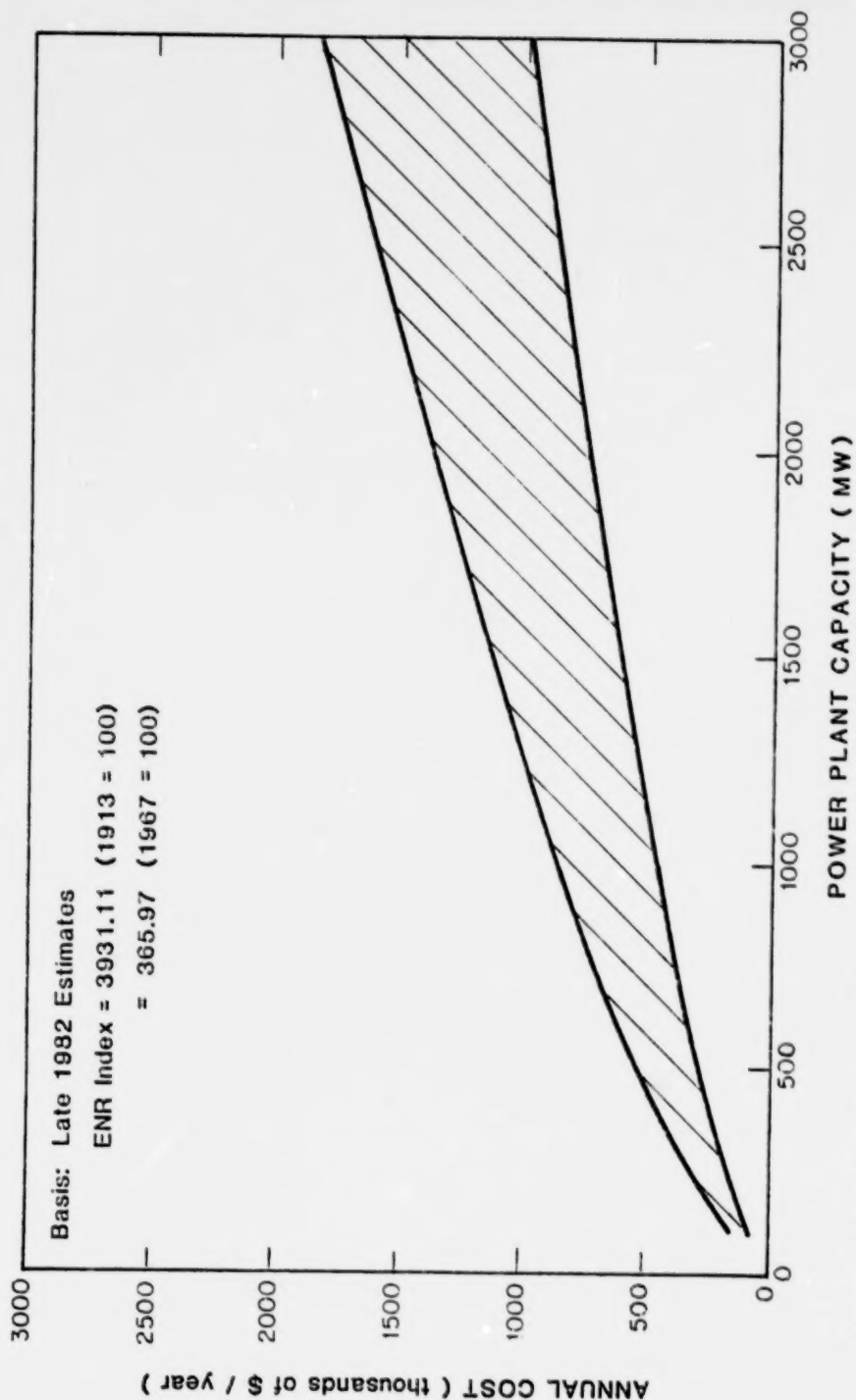


FIGURE 6.26 ANNUAL COST VERSUS POWER PLANT CAPACITY  
 FLY ASH HANDLING and PROCESSING  
 ( Wet Handling Without Recycle )

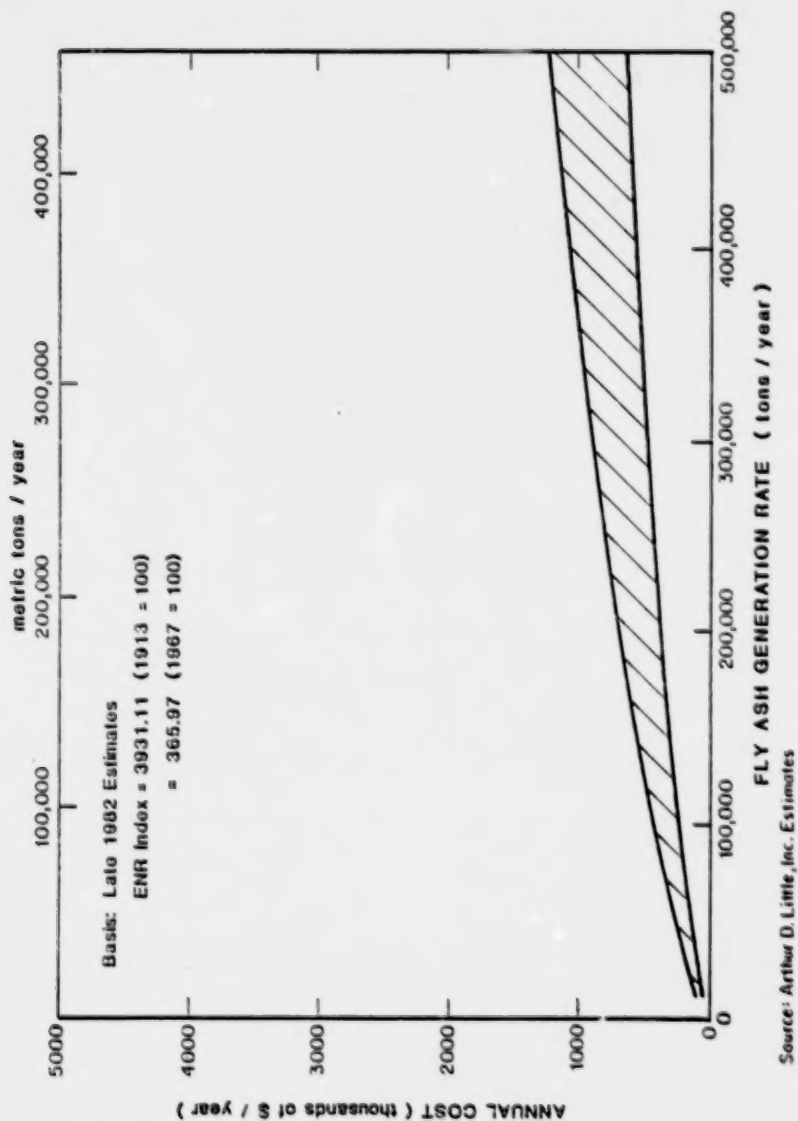
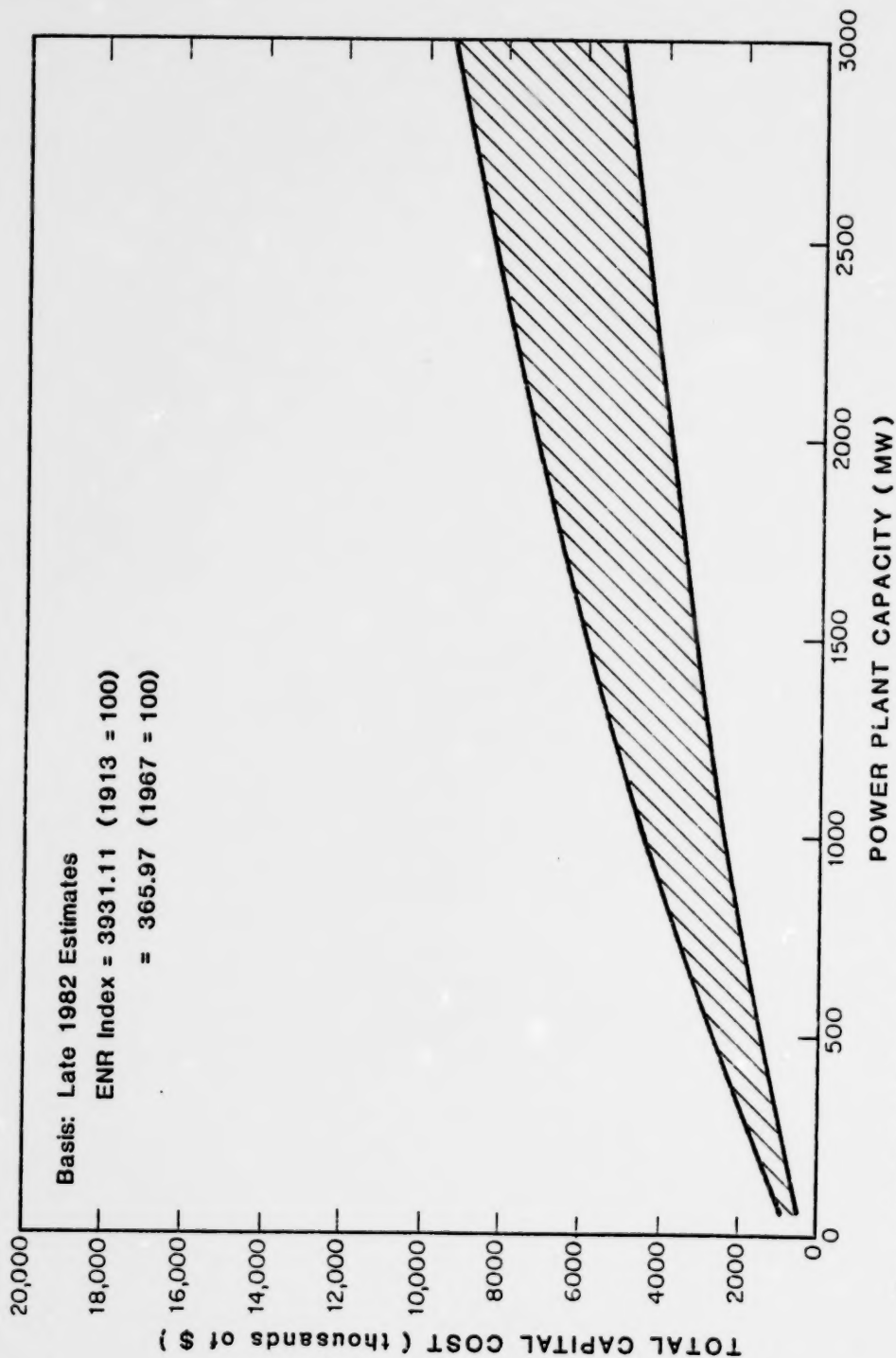


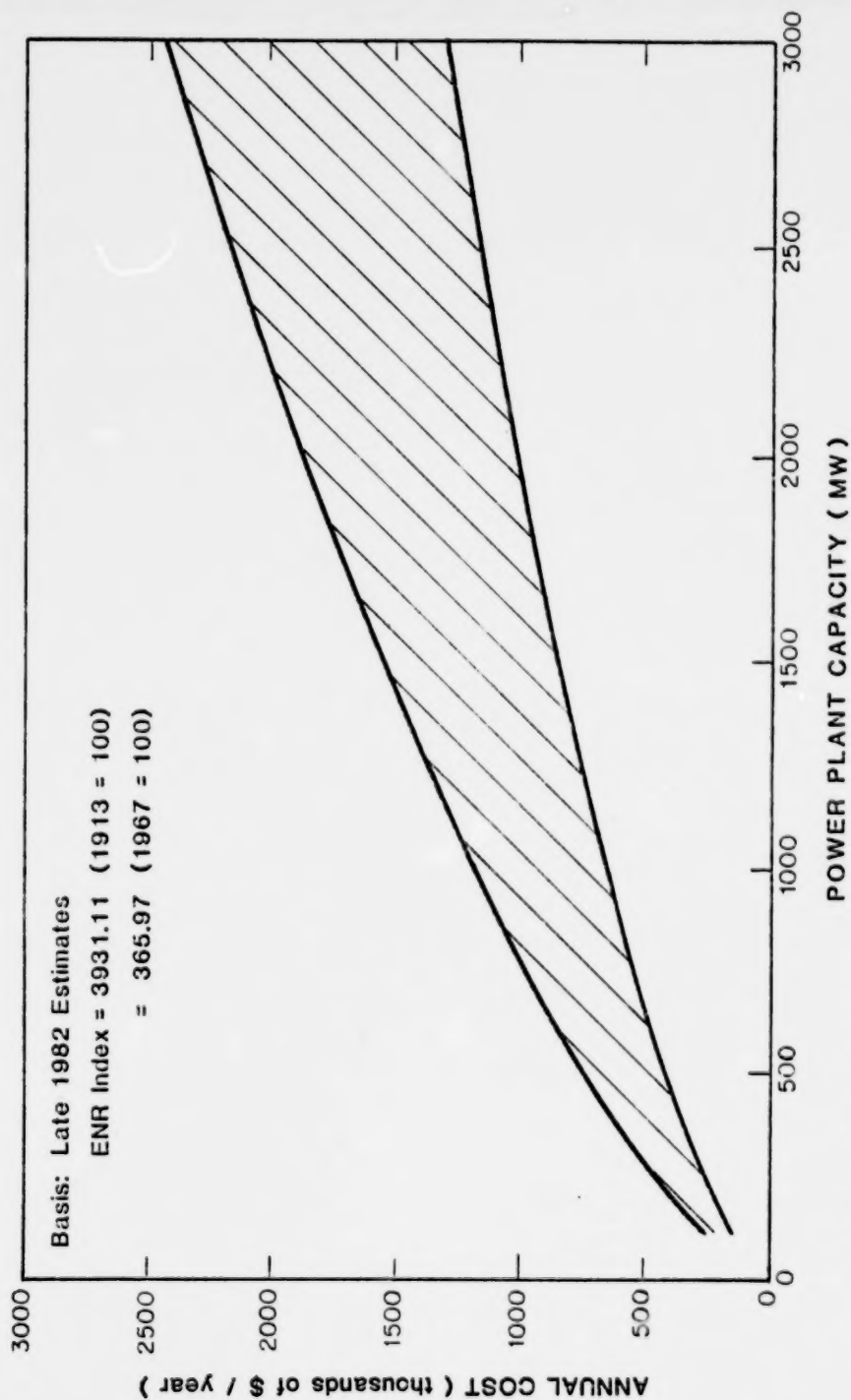
**FIGURE 6.27 ANNUAL COST VERSUS FLY ASH GENERATION RATE**  
**FLY ASH HANDLING AND PROCESSING**  
**( Wet Handling Without Recycle )**



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.28 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY**  
**FLY ASH HANDLING and PROCESSING**  
**( Wet Handling With Recycle )**





Source: Arthur D. Little, Inc. Estimates

FIGURE 6.29 ANNUAL COST VERSUS POWER PLANT CAPACITY  
FLY ASH HANDLING AND PROCESSING  
( Wet Handling With Recycle )

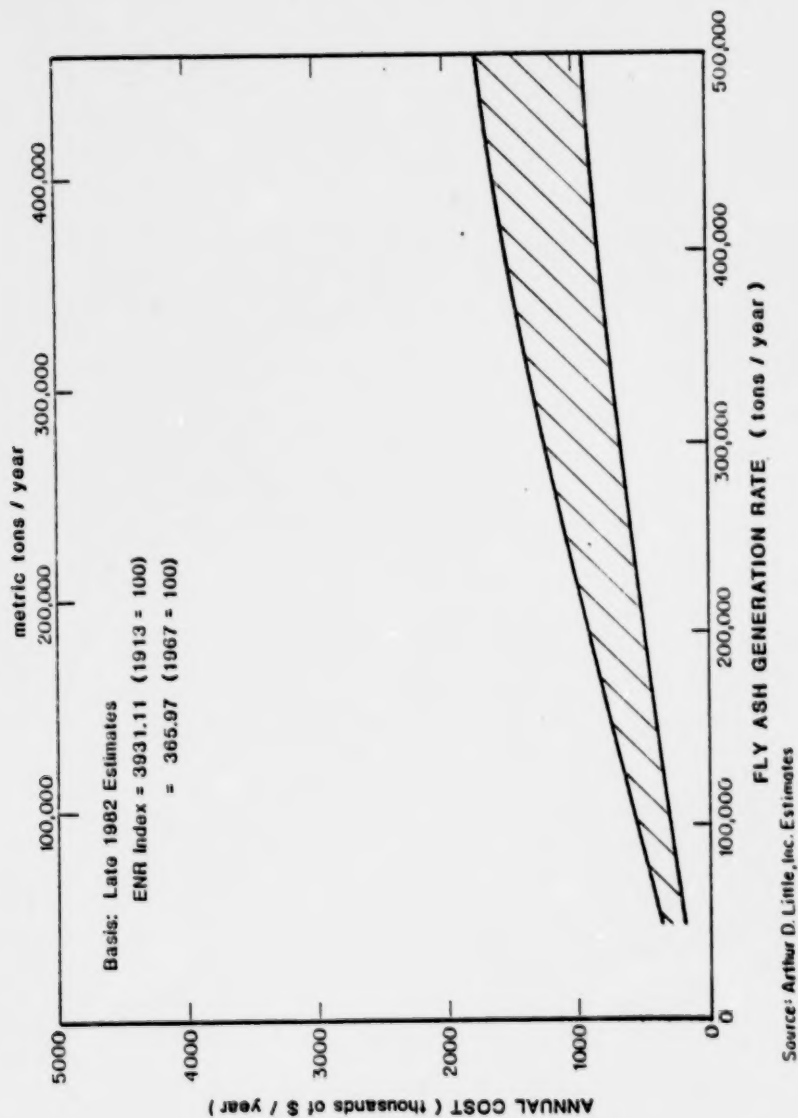


FIGURE 6.30 ANNUAL COST VERSUS FLY ASH GENERATION RATE  
FLY ASH HANDLING AND PROCESSING  
( Wet Handling With Recycle )

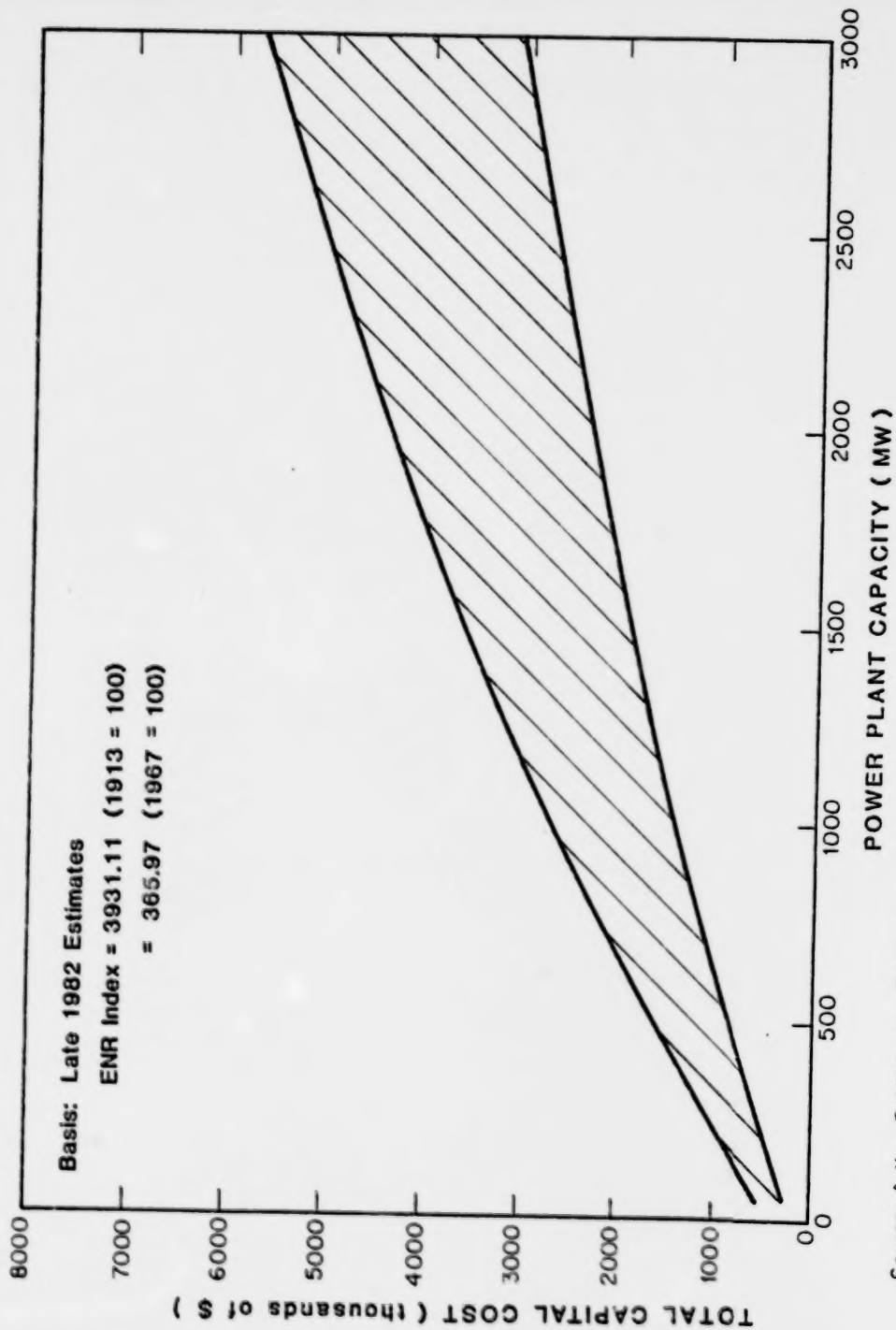
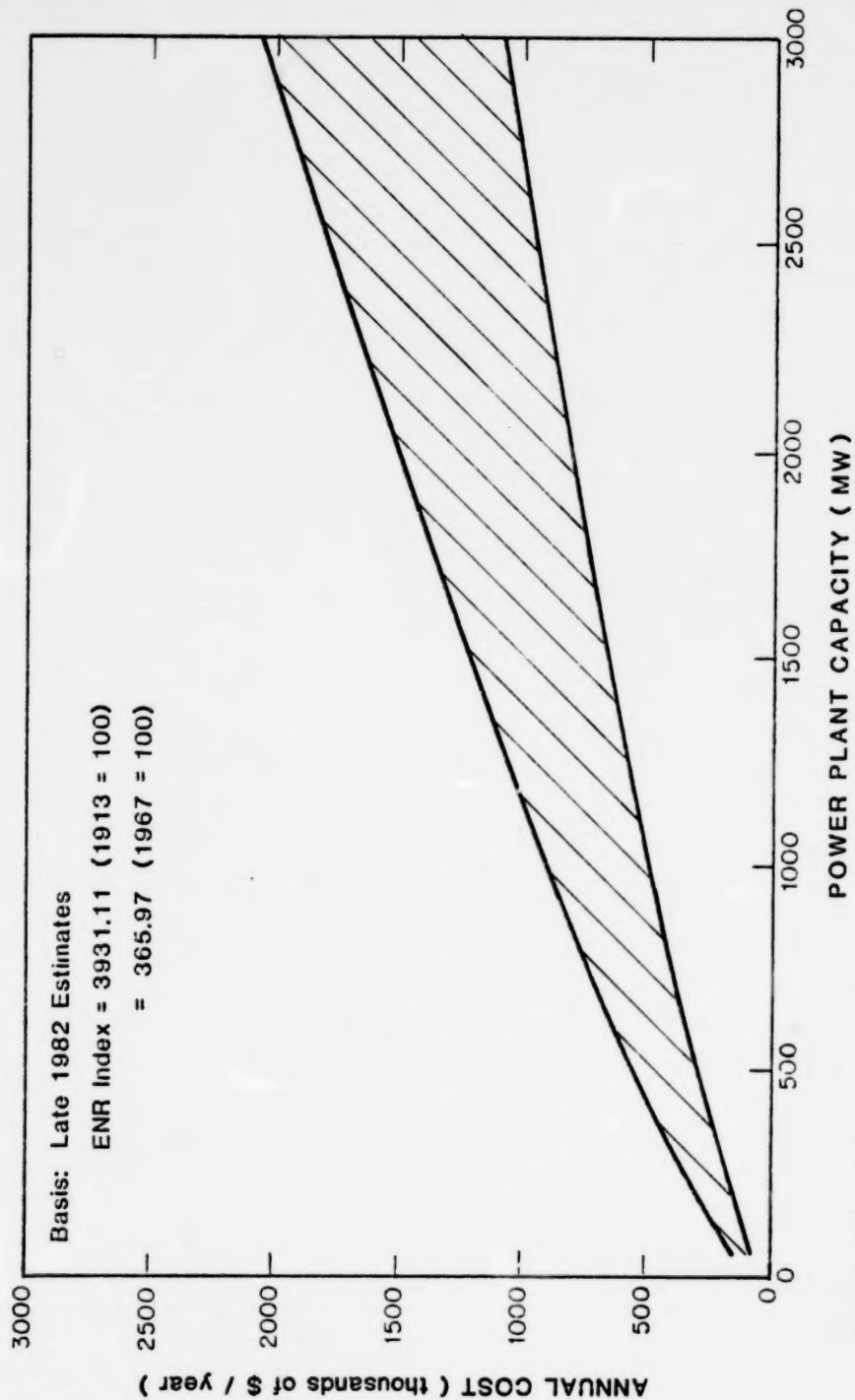
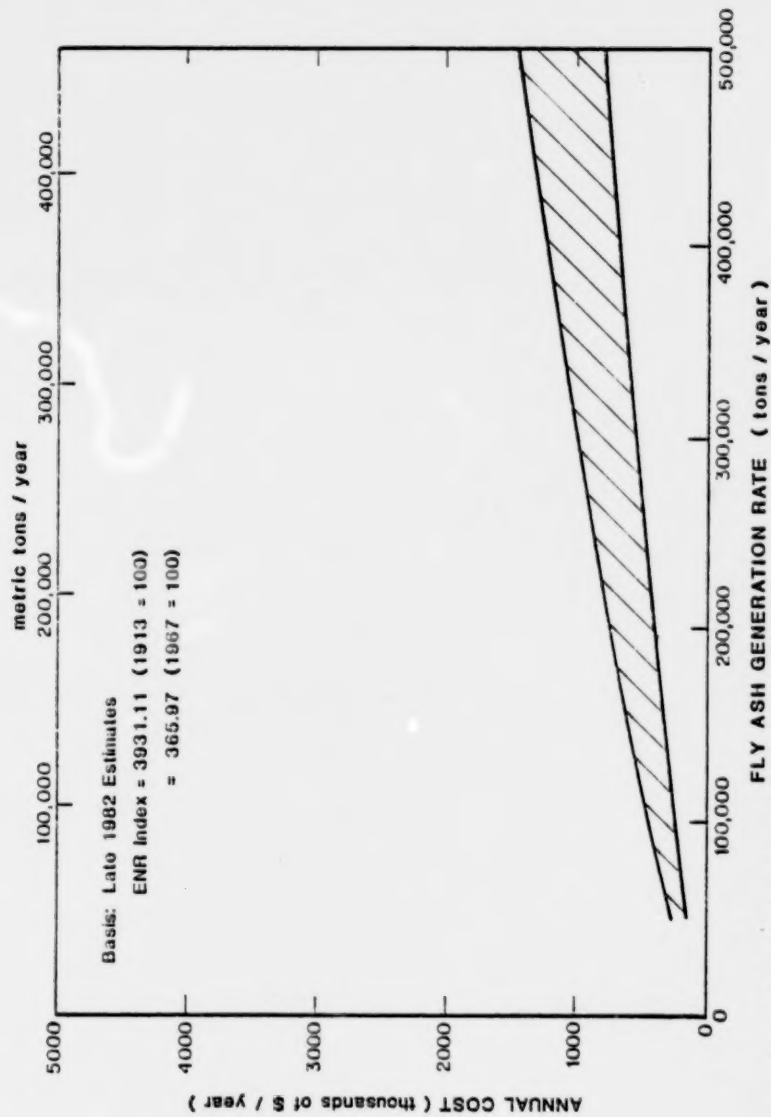


**FIGURE 6.31 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY**  
**FLY ASH HANDLING AND PROCESSING**  
**( Dry Handling )**



Source: Arthur D. Little, Inc. Estimates

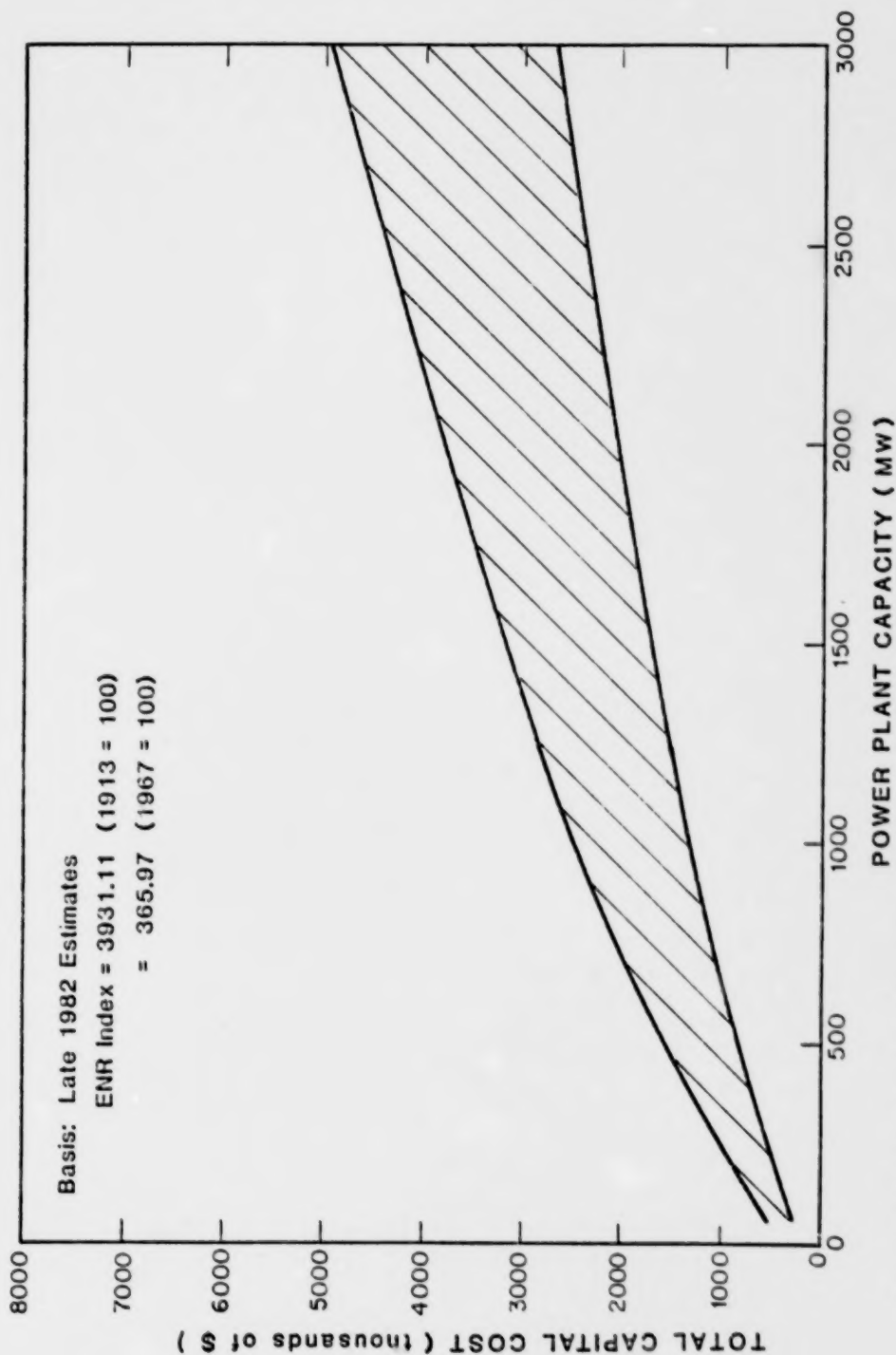
**FIGURE 6.32 ANNUAL COST VERSUS POWER PLANT CAPACITY  
 FLY ASH HANDLING and PROCESSING  
 ( Dry Handling )**



Source: Arthur D. Little, Inc. Estimates

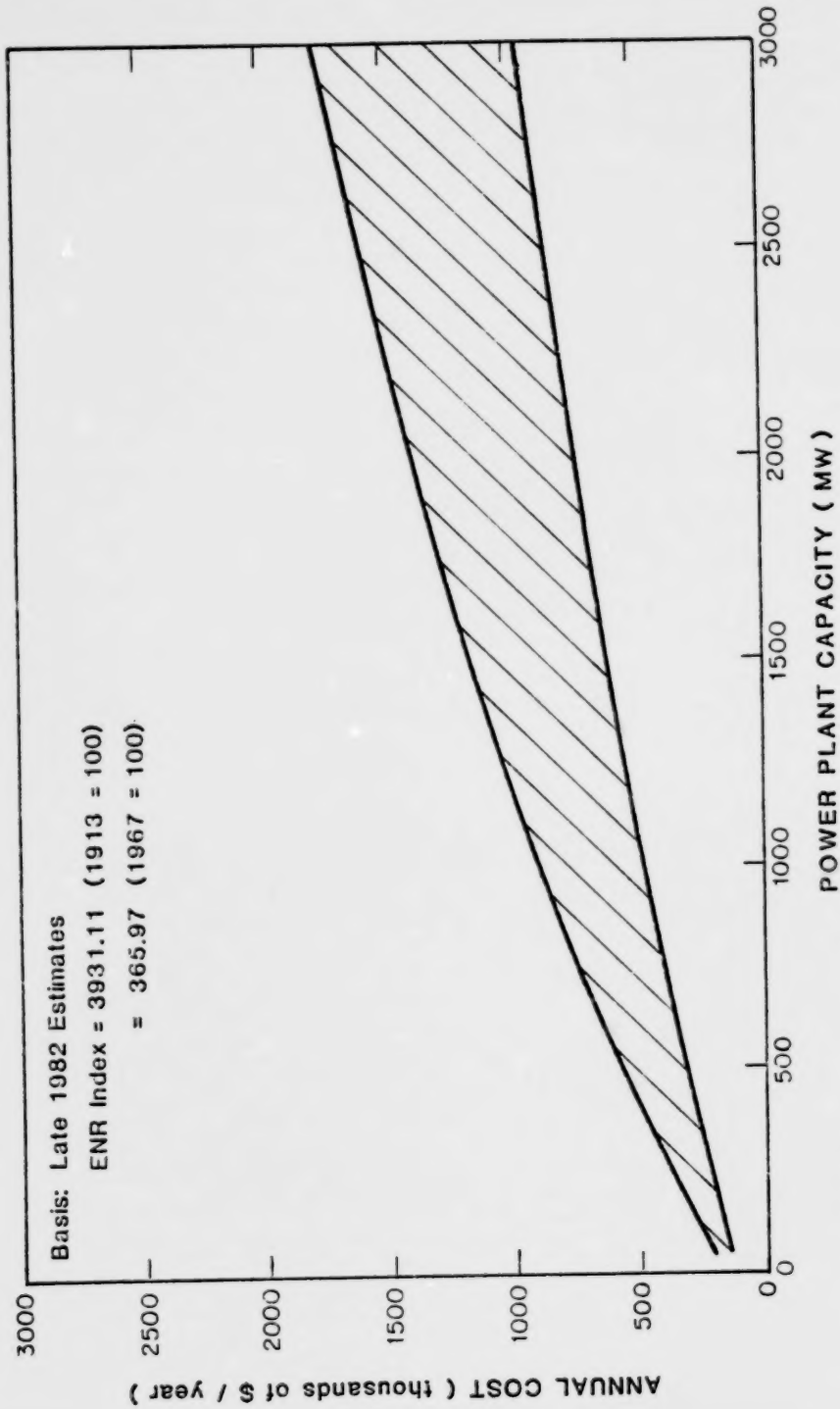
FIGURE 6.33 ANNUAL COST VERSUS FLY ASH GENERATION RATE  
 FLY ASH HANDLING AND PROCESSING  
 ( Dry Handling )





Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.34 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
 BOTTOM ASH HANDLING and PROCESSING  
 ( Wet Handling Without Recycle )**



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.35 ANNUAL COST VERSUS POWER PLANT CAPACITY**  
**BOTTOM ASH HANDLING and PROCESSING**  
 ( Wet Handling Without Recycle )

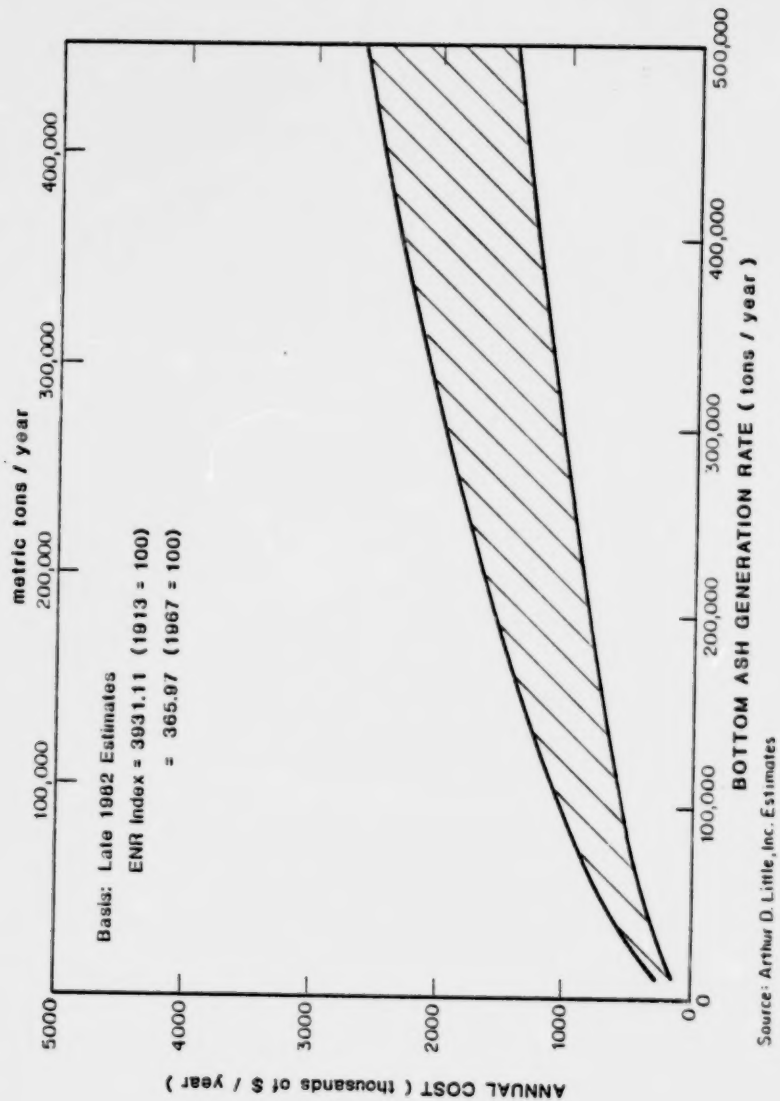


FIGURE 6.36 ANNUAL COST VERSUS BOTTOM ASH GENERATION RATE  
 BOTTOM ASH HANDLING and PROCESSING  
 ( Wet Handling Without Recycle )

considered, while the other option [represented in Figures 6.37 (capital costs), 6.38 and 6.39 (annual costs)] refers to wet bottom ash handling systems which incorporate supernatant recycle provisions. In both cases, cost variations are dictated by the use or absence of bottom ash dewatering. The use of interim ponds for dewatering was not considered in preparing the bottom ash handling and processing cost curves, due to a paucity of available interim pond cost data (see Fly Ash Handling and Processing). Thus, for bottom ash handling and processing, the only dewatering alternative considered with respect to the development of cost curves was the use of hydrobins (i.e., dewatering bins). Bottom ash handling and processing costs for systems incorporating hydrobins will typically fall in the higher ranges of the cost band than those for simple wet handling systems which are used in conjunction with sluice pipeline transport systems.

FGD Waste Handling and Processing -- Capital and annual costs developed for FGD waste handling and processing are presented in Figures 6.40 and 6.41, respectively. Costs for FGD waste handling and processing varies according to the degree of processing employed, which ranges from no processing to dewatering and finally to dewatering with stabilization or chemical fixation. The lower ranges of the capital and annual cost curve bands correspond to costs for FGD waste handling with limited or no processing. The upper region of the band relates to extensive processing (i.e., primary and secondary dewatering of the waste followed by chemical fixation). The intermediate cost curve regions generally pertain to systems which incorporate only dewatering for FGD waste processing. In this case, as with fly and bottom ash wastes, the cost of interim ponding was not considered in developing the cost curves. To date, the use of interim ponds for the purpose of dewatering FGD wastes is very limited; in addition, it is not anticipated to grow in the future.

Fly Ash Storage -- Fly ash storage capital costs are dominated by the cost of the storage structure (i.e., the cost of the silo, bin, etc). The major cost variables with respect to storage structures are: (1) the size and number of structures required and (2) the material of construction. The cost curves provided for fly ash storage are based on the premise that the number of storage vessels and vessel sizes are optimized with respect to cost. Fly ash storage in structures constructed of relatively lower cost materials, such as carbon steel, would be expected to have relatively lower costs (and, hence, would fall in the lower ranges of the cost curve band) than those constructed of more costly materials, such as rubber-lined steel. The choice of materials of construction are site specific, depending on the chemical properties of the ash to be stored. The cost curves for fly ash storage are presented in Figures 6.42 (capital costs), 6.43 and 6.44 (annual costs).

Raw Materials Handling and Storage -- The capital and annual cost curves for the raw materials handling and storage module are provided in Figures 6.45 and 6.46, respectively. The upper ranges of the cost curves pertain to raw materials handling and storage associated with CSI or Dravo chemical fixation processes where multiple additives, and therefore multiple storage

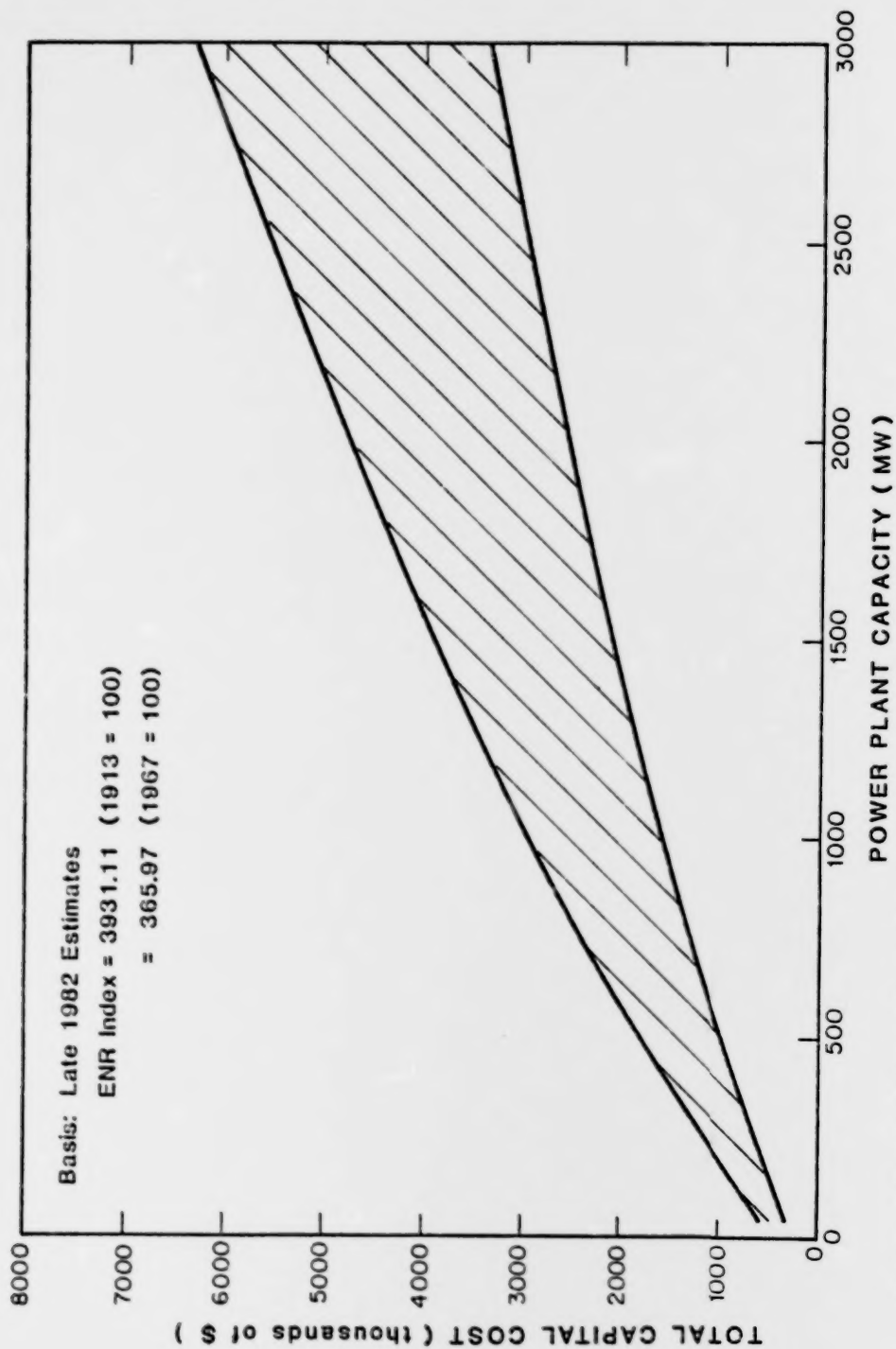
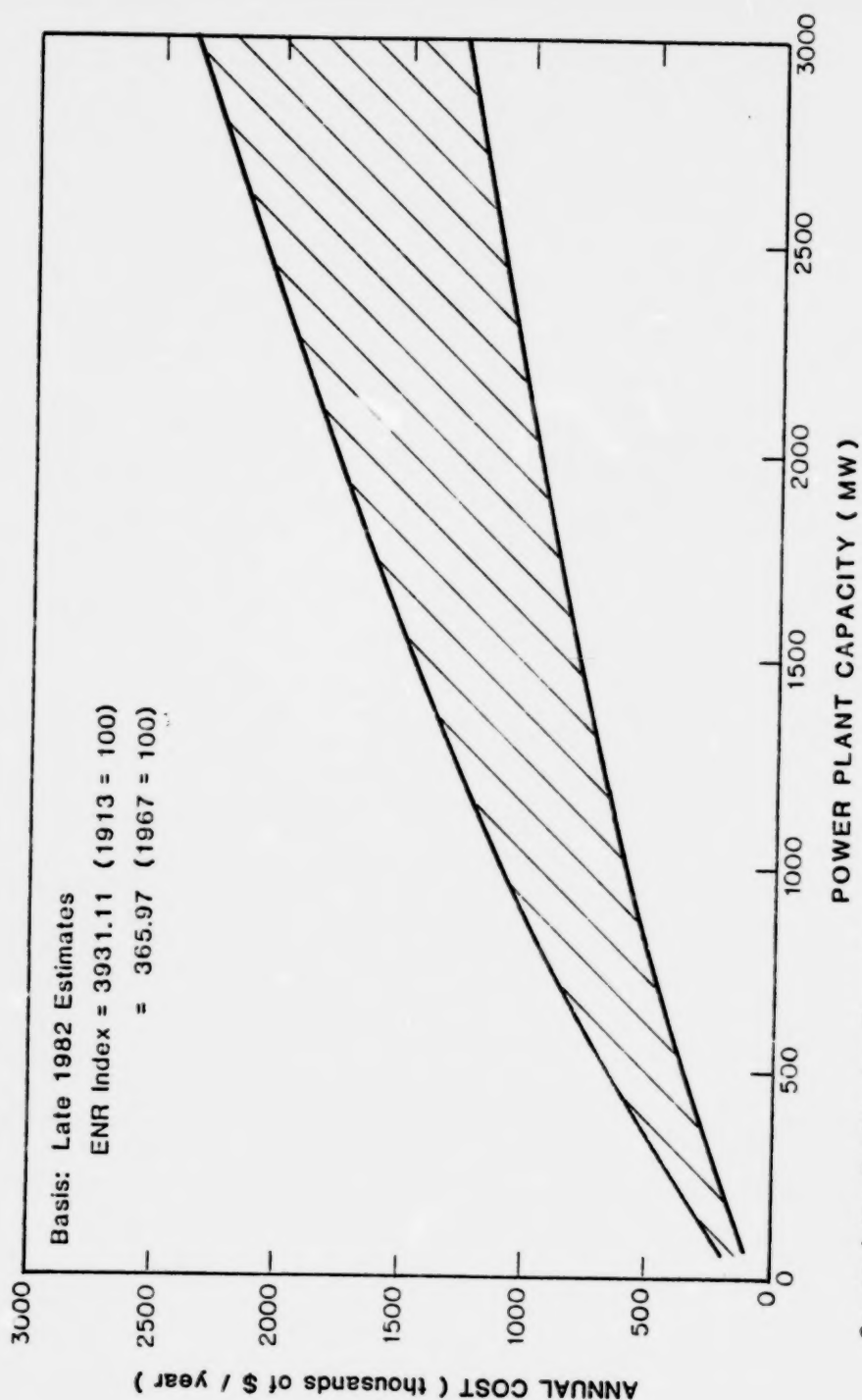


**FIGURE 6.37 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
 BOTTOM ASH HANDLING and PROCESSING  
 ( Wet Handling With Recycle )**



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.38 ANNUAL COST VERSUS POWER PLANT CAPACITY**  
 BOTTOM ASH HANDLING and PROCESSING  
 ( Wet Handling With Recycle )



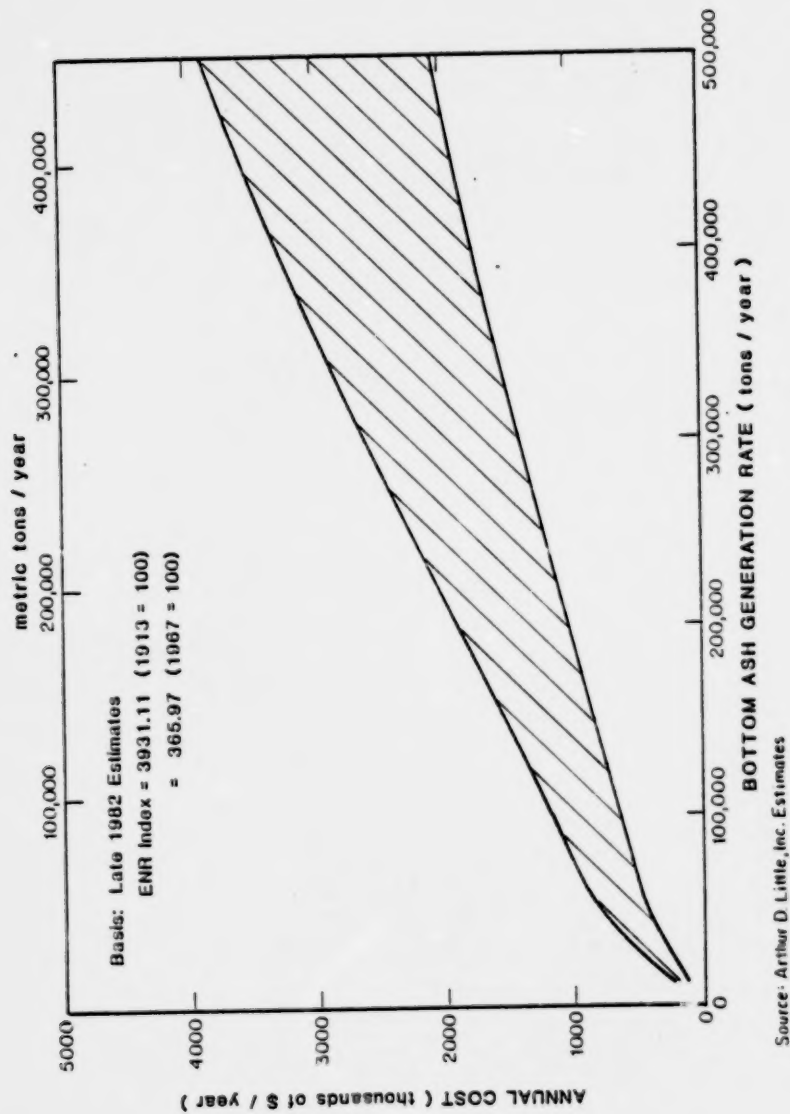


FIGURE 6.39 ANNUAL COST VERSUS BOTTOM ASH GENERATION RATE  
BOTTOM ASH HANDLING AND PROCESSING  
( Wet Handling With Recycle )

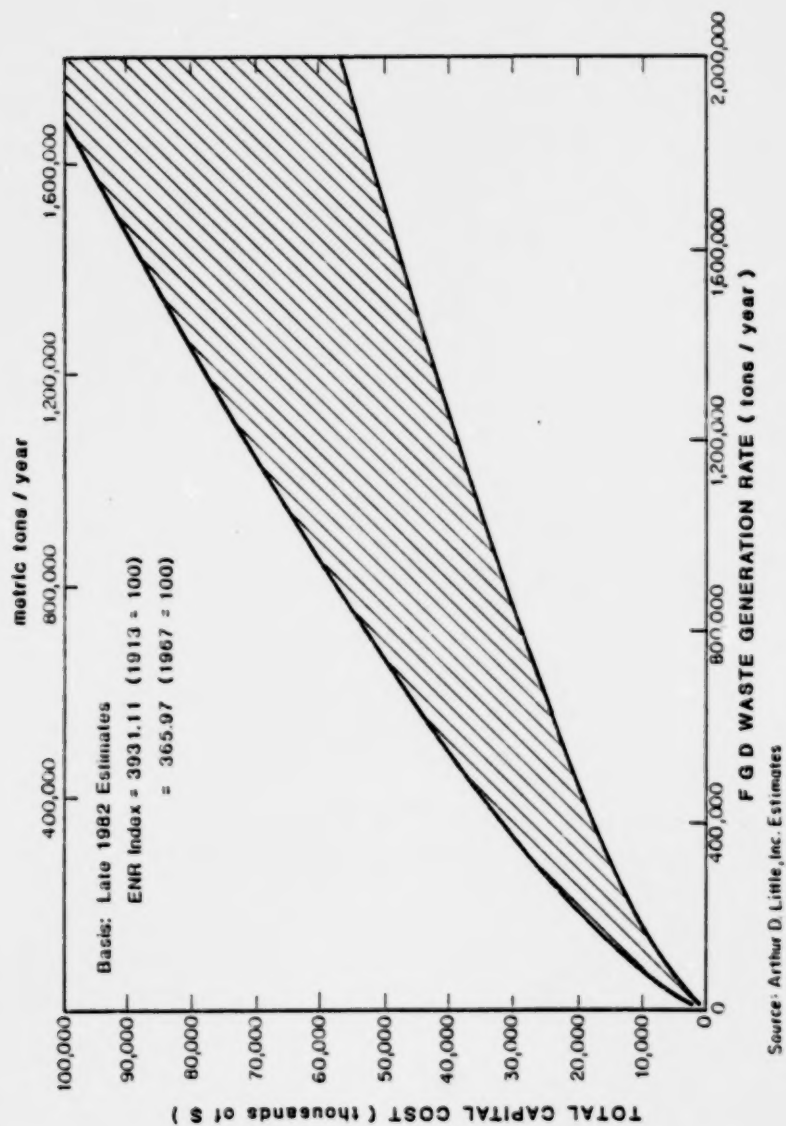


FIGURE 6.40 TOTAL CAPITAL COST VERSUS FGD WASTE GENERATION RATE  
FGD WASTE HANDLING AND PROCESSING

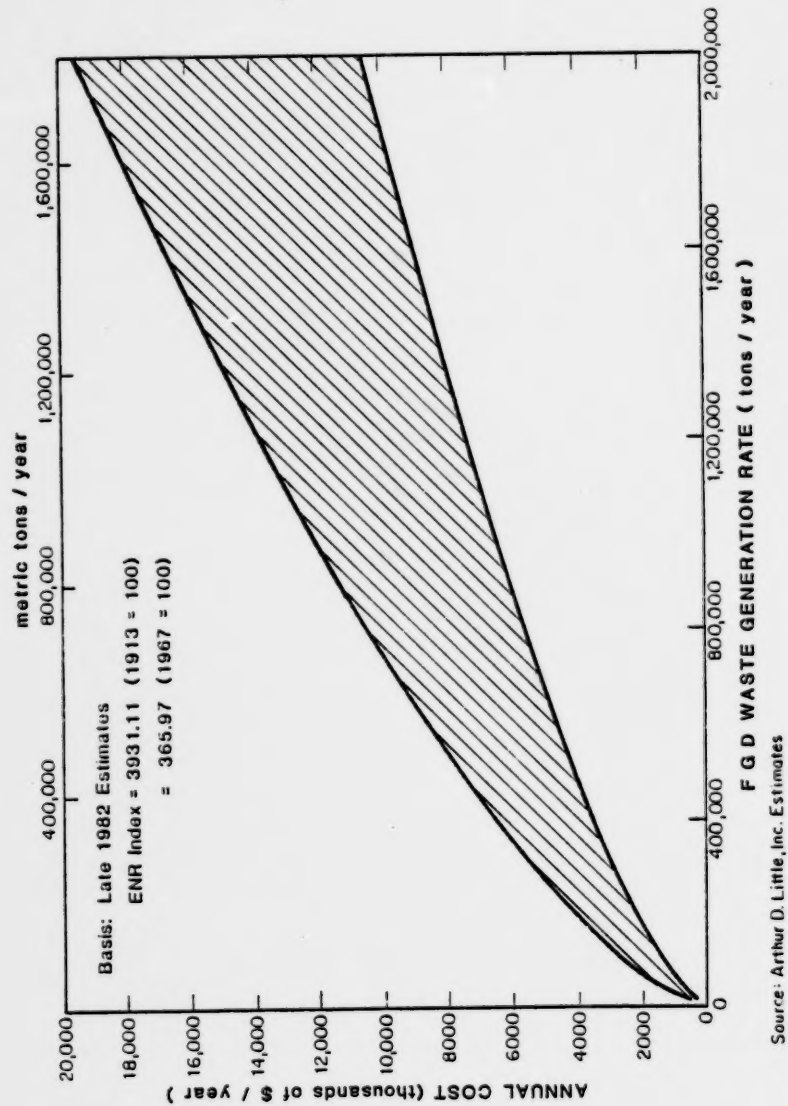
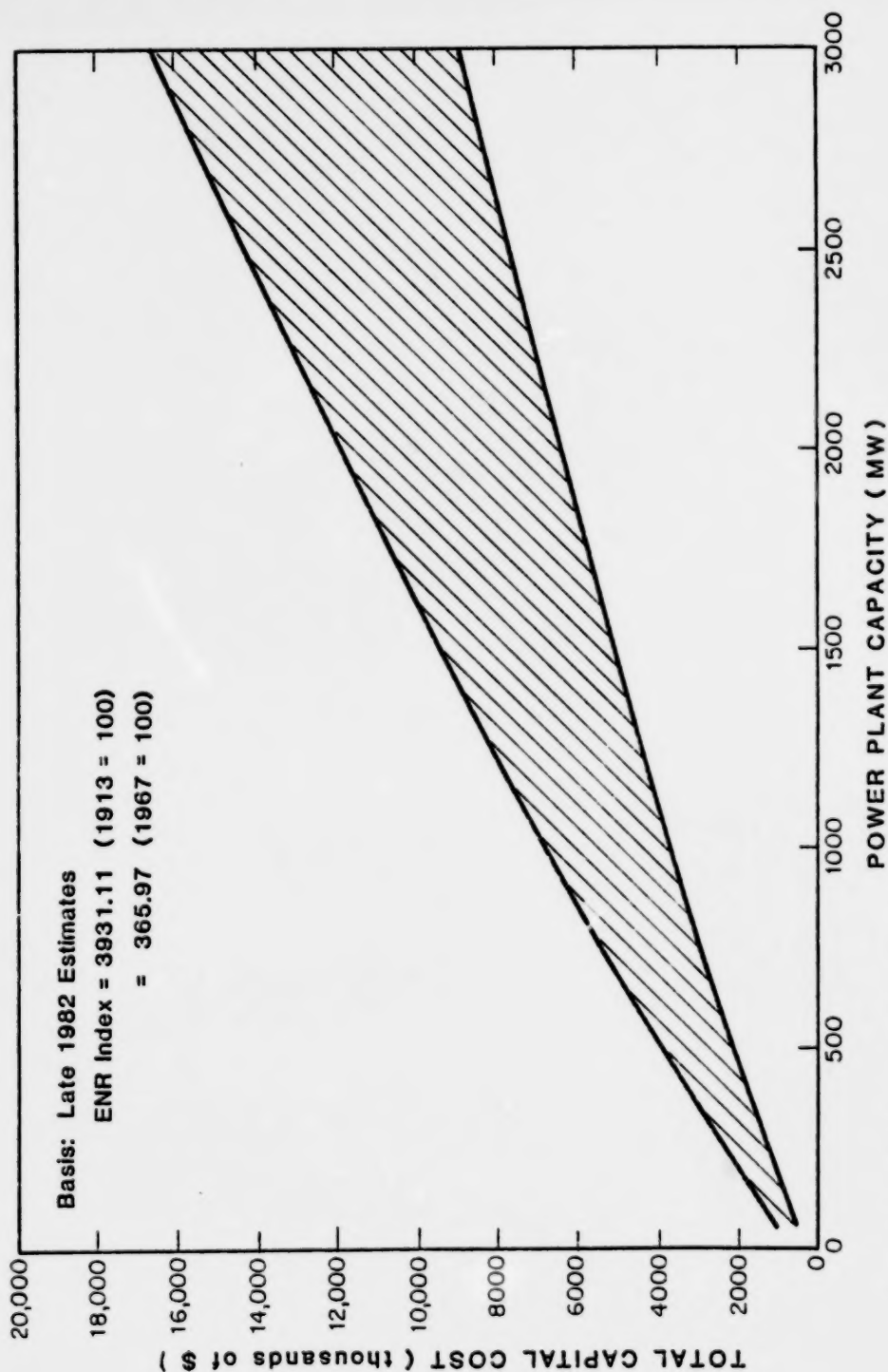
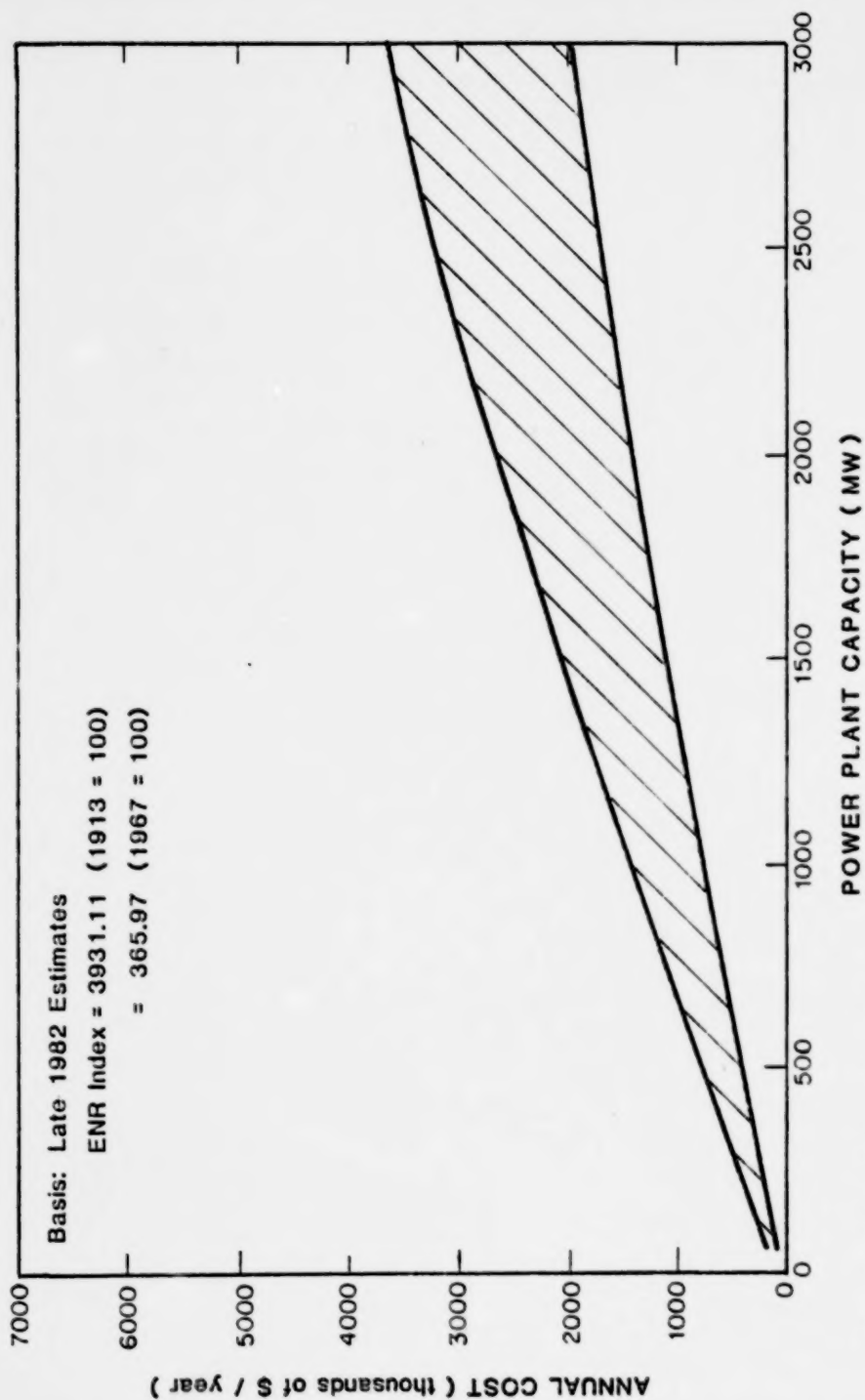


FIGURE 6.41 ANNUAL COST VERSUS FGD WASTE GENERATION RATE  
 FGD WASTE HANDLING AND PROCESSING



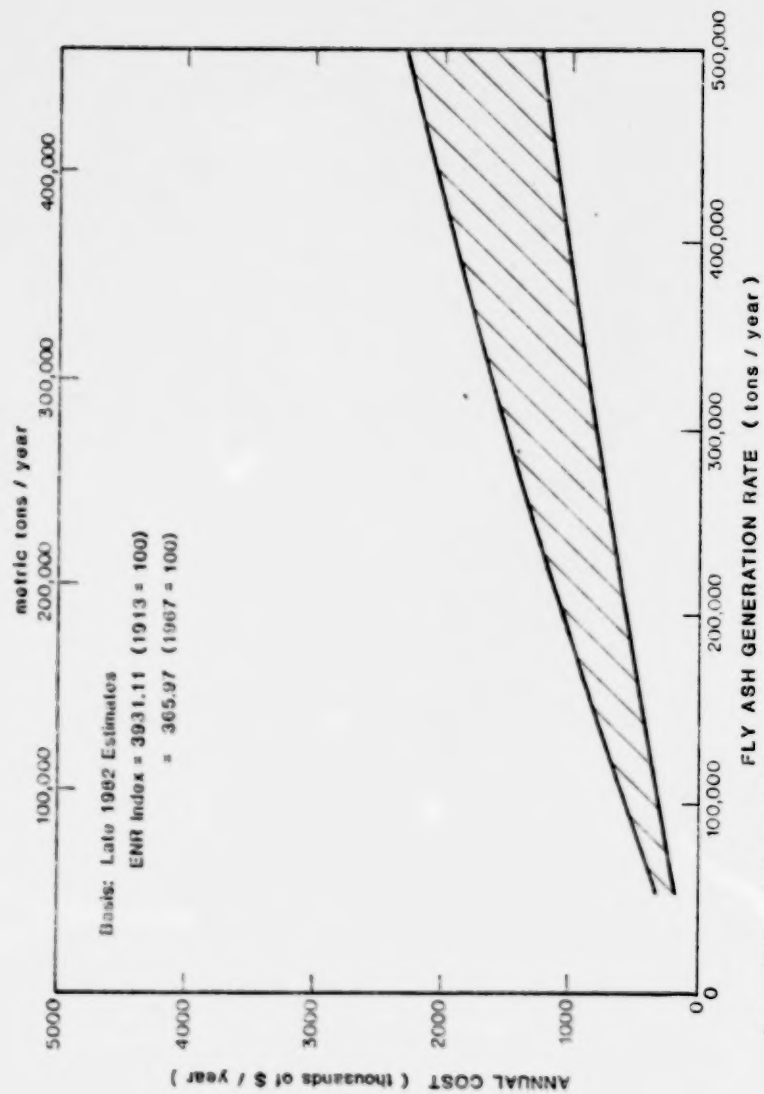
Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.42 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
FLY ASH STORAGE**



Source: Arthur D. Little, Inc. Estimates

FIGURE 6.43 ANNUAL COST VERSUS POWER PLANT CAPACITY  
FLY ASH STORAGE



Source: Arthur D. Little, Inc. Estimates

FIGURE 6.44 ANNUAL COST VERSUS FLY ASH GENERATION RATE  
 FLY ASH STORAGE



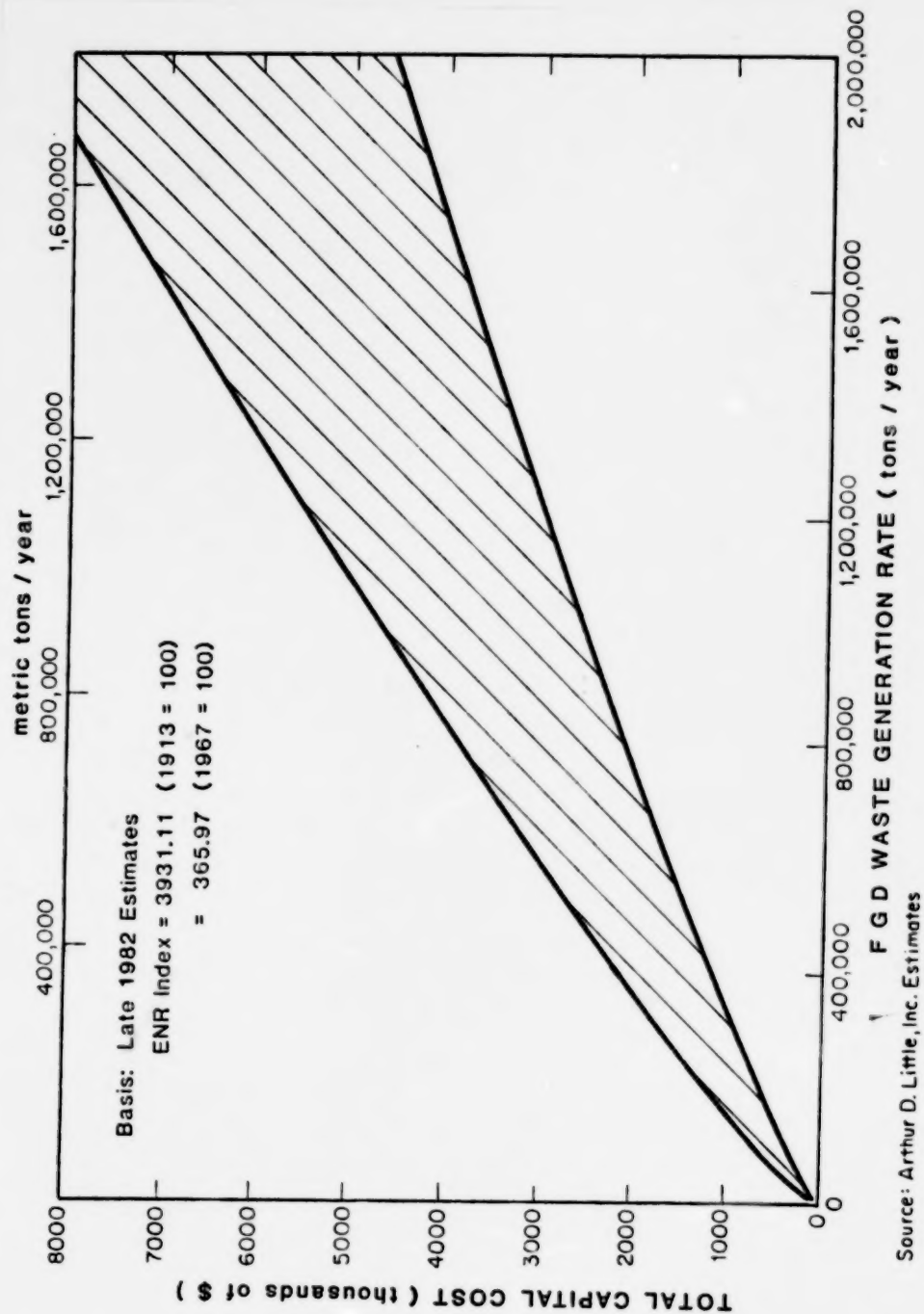


FIGURE 6.45 TOTAL CAPITAL COST VERSUS F G D WASTE GENERATION RATE  
 RAW MATERIALS HANDLING AND STORAGE

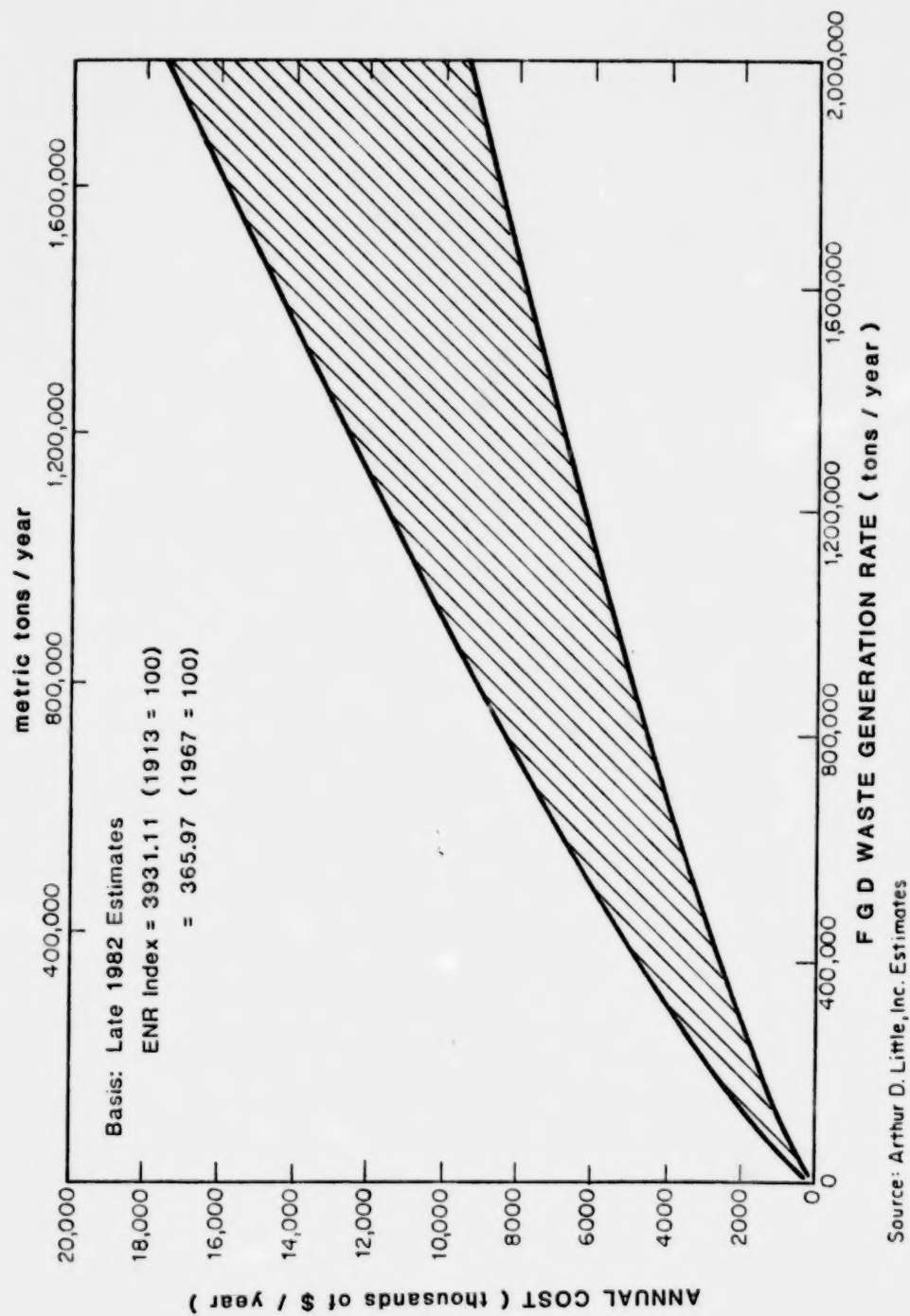


**FIGURE 6.46 ANNUAL COST VERSUS FGD WASTE GENERATION RATE  
 RAW MATERIALS HANDLING AND STORAGE**

systems, (potentially with some common elements) are required. For FGD waste stabilization processes (i.e., fly ash blending) where only one additive is used, the lower ranges of the cost curves apply.

Coal Ash and FGD Waste Transport -- Two types of transport options are available for coal-fired utility solid wastes, wet and dry systems. For this reason, two sets of capital and annual cost curves were developed for each of the three coal-fired utility waste types (fly ash, bottom ash and FGD waste). For each waste type, capital and annual cost curves were developed for wet transport by pipelines; a second set of curves pertaining to truck transport of dry wastes was also developed.

The transport modules for each of the three waste types show little variation with respect to design parameters that affect costs. Thus, the following discussion pertains to all three types of coal-fired utility solid wastes.

Capital and annual costs for pipeline transport vary according to the following parameters:

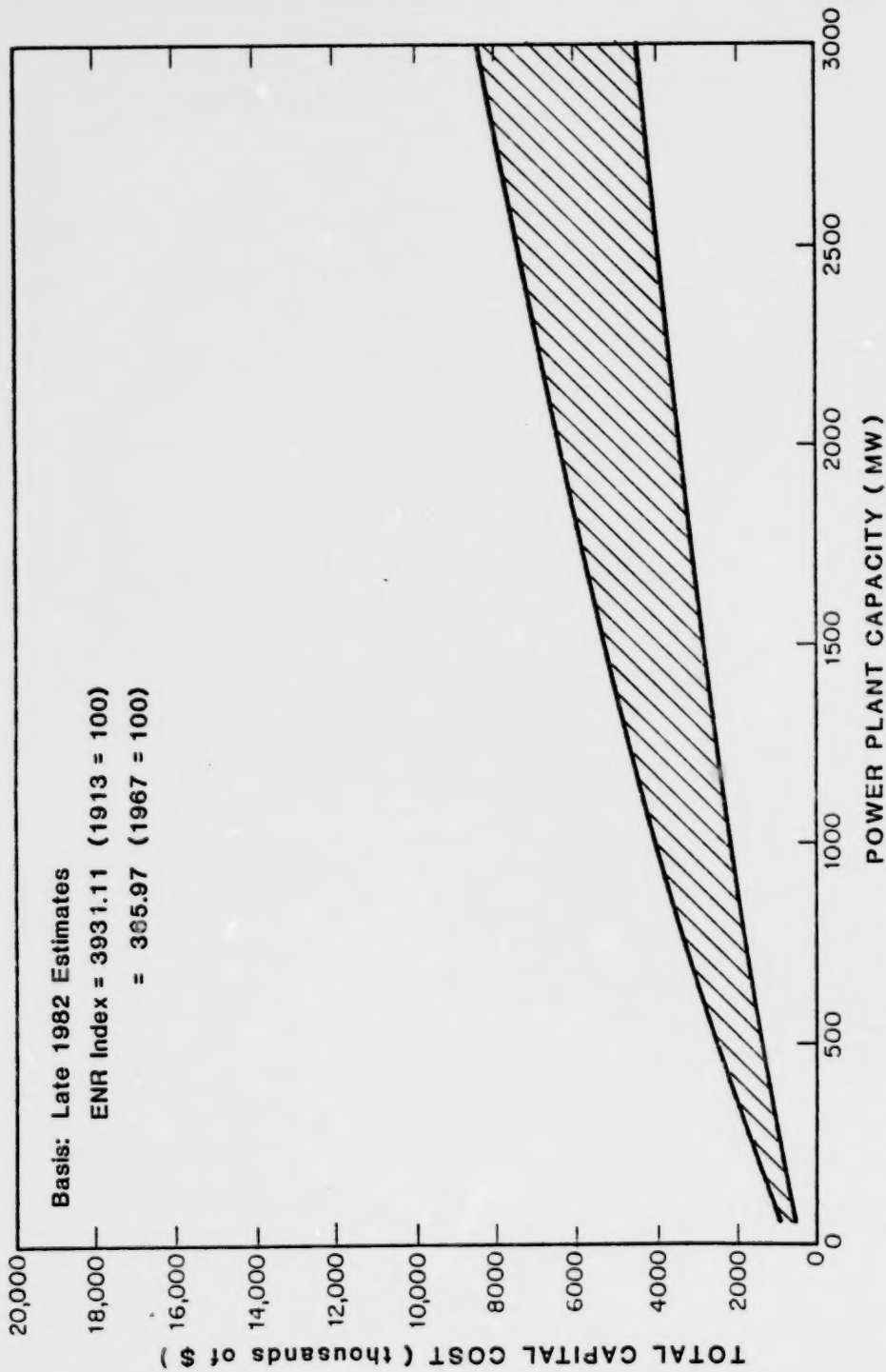
- transport distance;
- pipeline material of construction; and
- terrain.

The cost curves presented for pipeline transport of fly ash (Figure 6.47 for capital costs and Figures 6.48 and 6.49 for annual costs), bottom ash (Figure 6.50 for capital costs and Figures 6.51 and 6.52 for annual costs) and FGD wastes (Figures 6.53 and 6.54 for capital and annual costs, respectively) do not consider variability of transport distance. A transport distance of 1.6 km (1 mile) was assumed for the purposes of this study. Of course, for different transport distances, costs will vary. At present, correction factors to account for such differences have not been developed (see Section 6.3) due to insufficient data.

Pipeline materials of construction vary according to the erosive and corrosive nature of the media transported. The selection of pipe materials is highly site-specific because of the wide range of waste characteristics. With respect to the capital and annual cost curves, pipelines constructed of lower cost materials (e.g., carbon steel) will exhibit costs in the lower regions of the cost band. The costs for pipelines constructed of more expensive materials (e.g., butyl rubber-lined steel, nickel/chrome alloy, fiberglass, etc.) will fall in the upper range of the cost curve band.

The effect of terrain on costs was not considered in this study; the terrain considered was assumed to be reasonably level.

Truck transport capital and operating cost curves were prepared for fly ash (Figure 6.55 for capital costs and Figures 6.56 and 6.57 for annual costs), bottom ash (Figure 6.58 for capital costs and Figures 6.59 and 6.60



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.47 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
 FLY ASH TRANSPORT ( Wet Sluicing )**

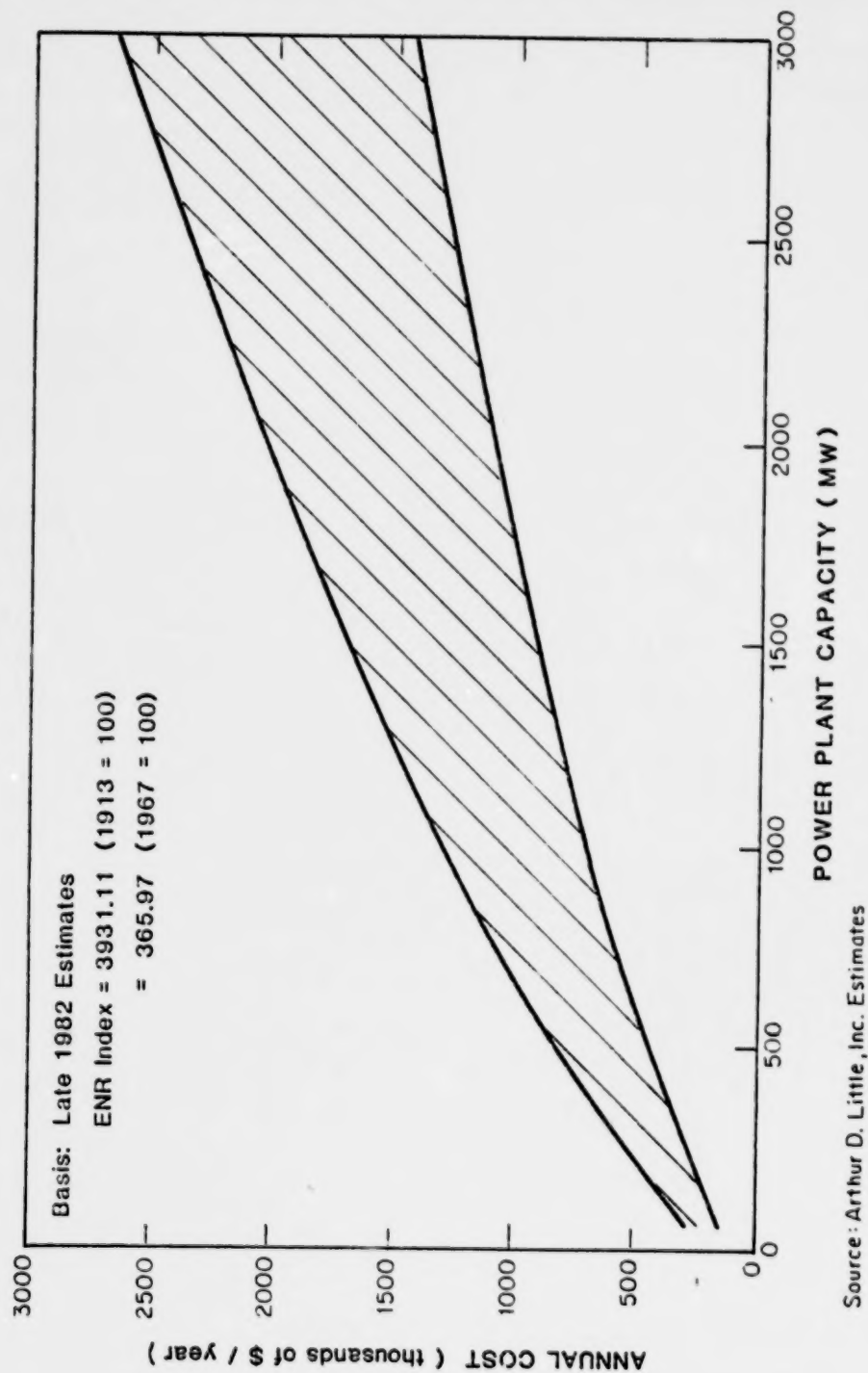


FIGURE 6.48 ANNUAL COST VERSUS POWER PLANT CAPACITY  
FLY ASH TRANSPORT ( Wet Sluicing )

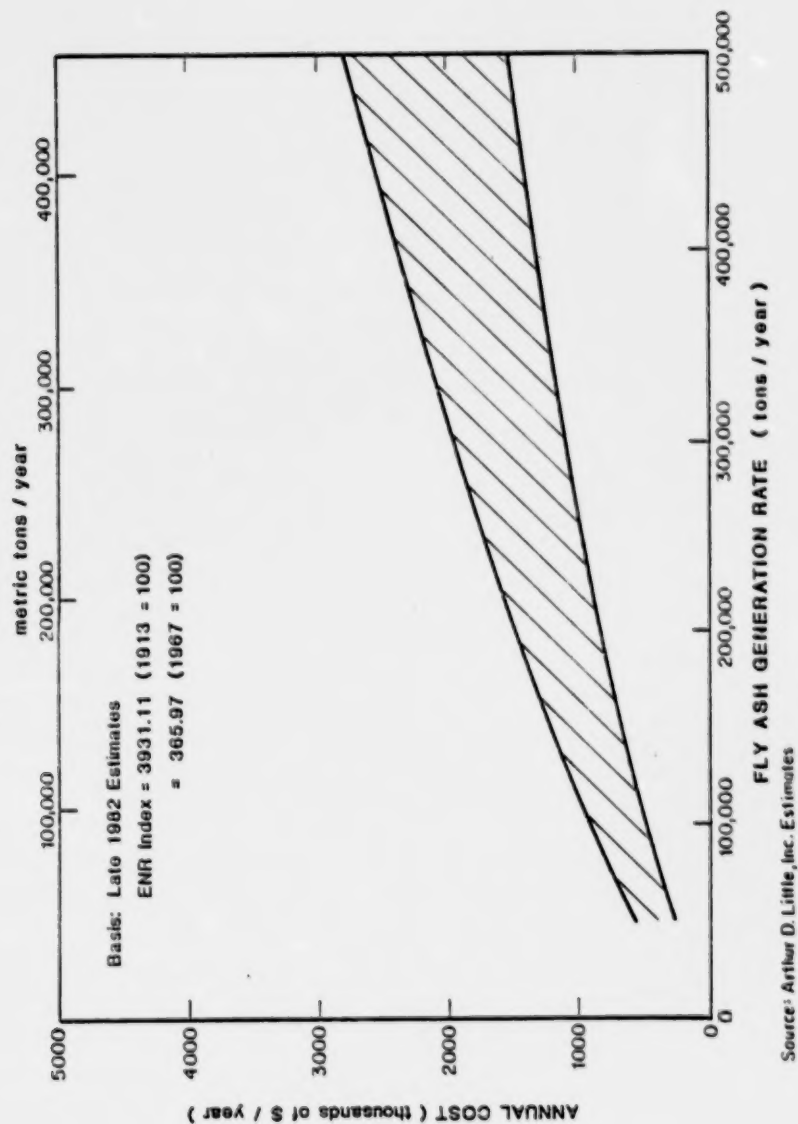
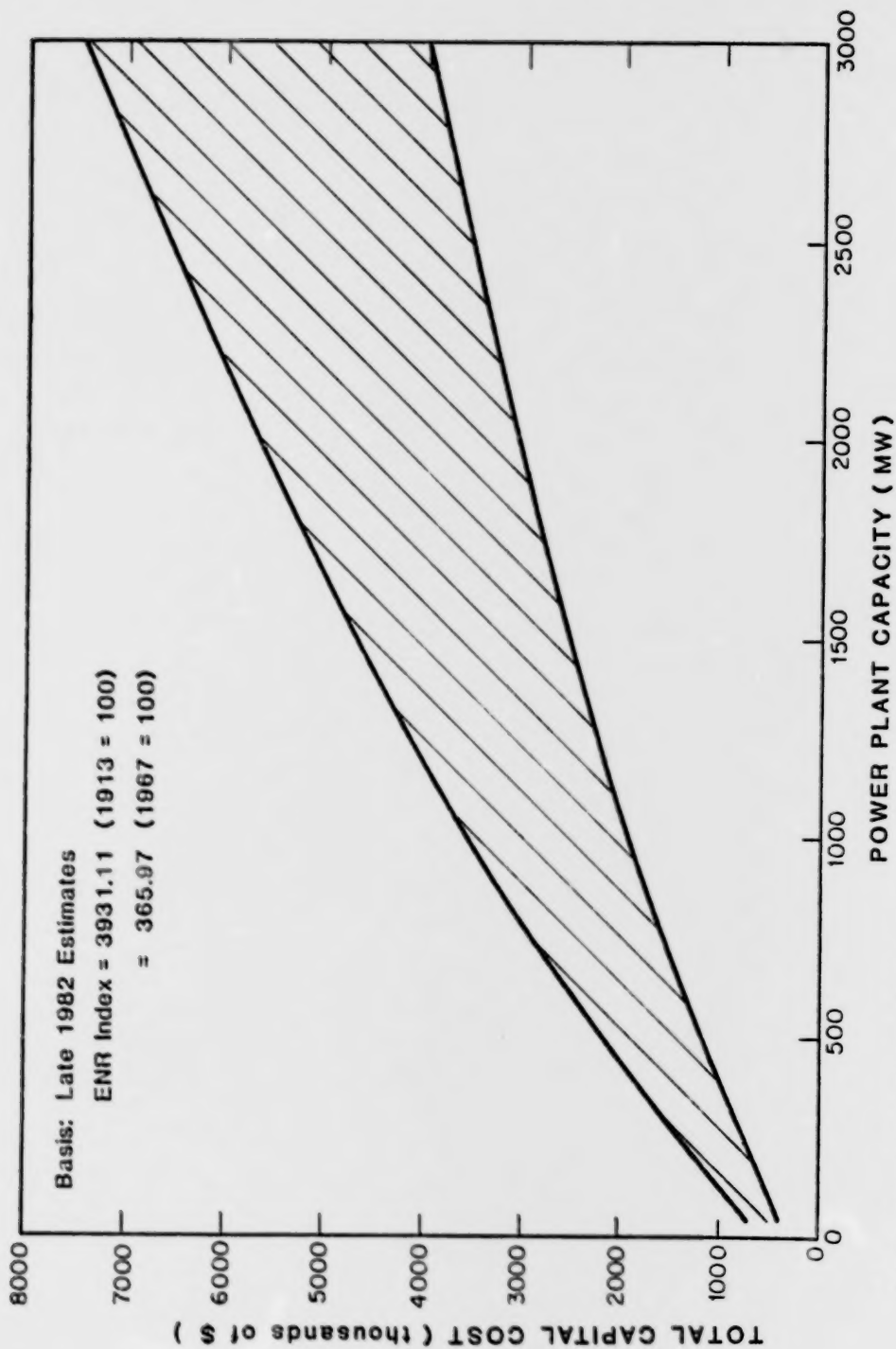


FIGURE 6.49 ANNUAL COST VERSUS FLY ASH GENERATION RATE  
FLY ASH TRANSPORT ( Wet Sluicing )





Source: Arthur D. Little, Inc. Estimates

FIGURE 6.50 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
 BOTTOM ASH TRANSPORT ( Wet Sluicing )

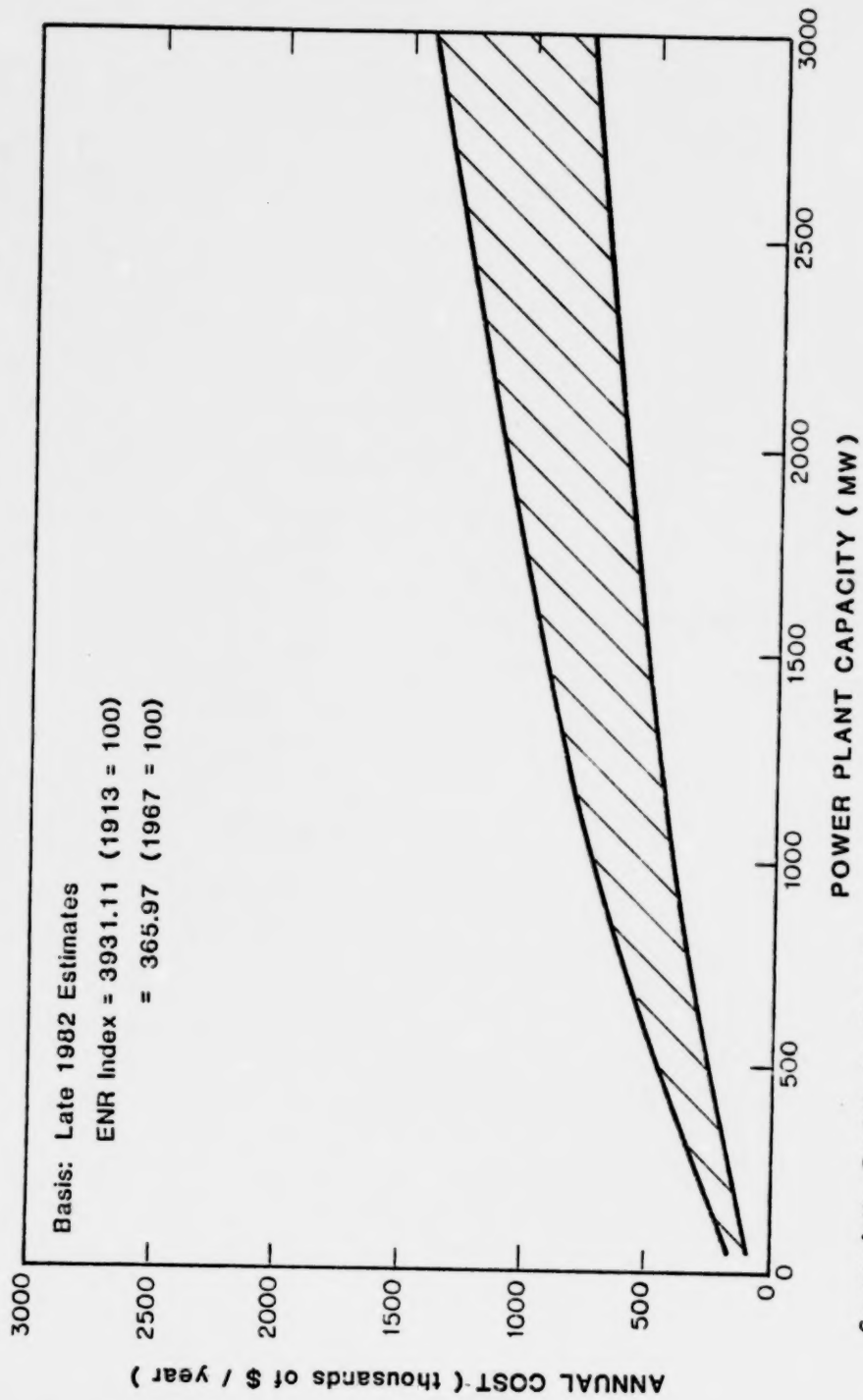


FIGURE 6.51 ANNUAL COST VERSUS POWER PLANT CAPACITY  
 BOTTOM ASH TRANSPORT ( Wet Sluicing )

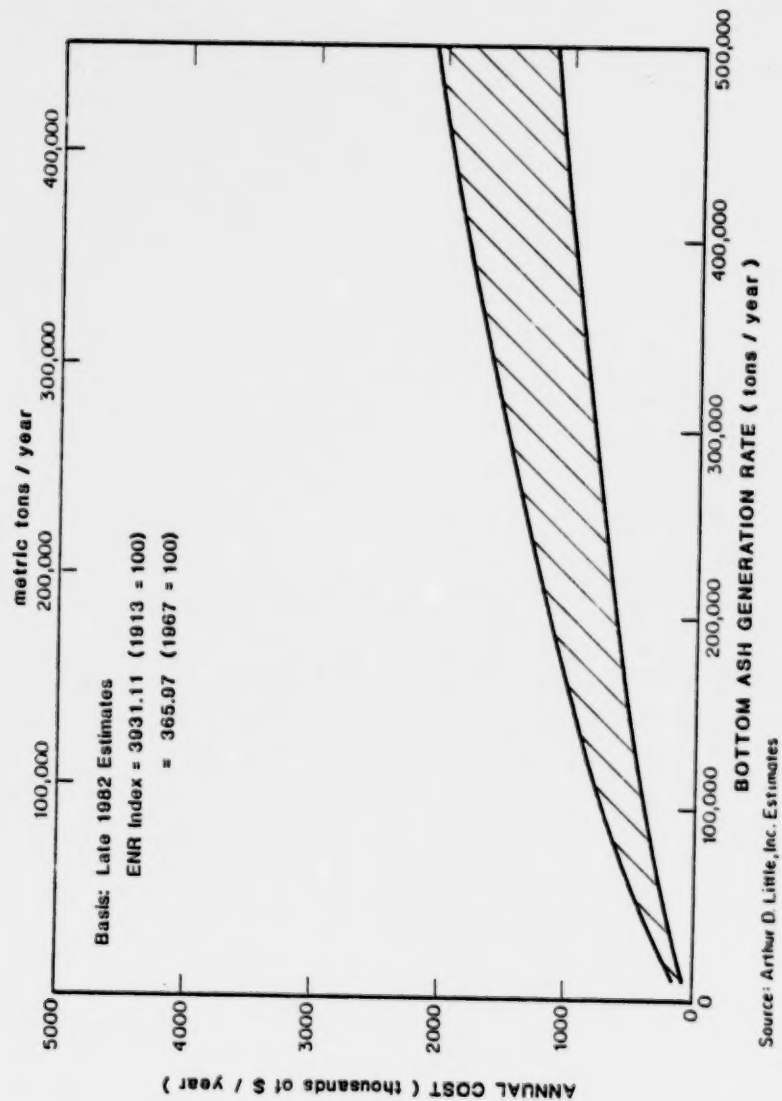
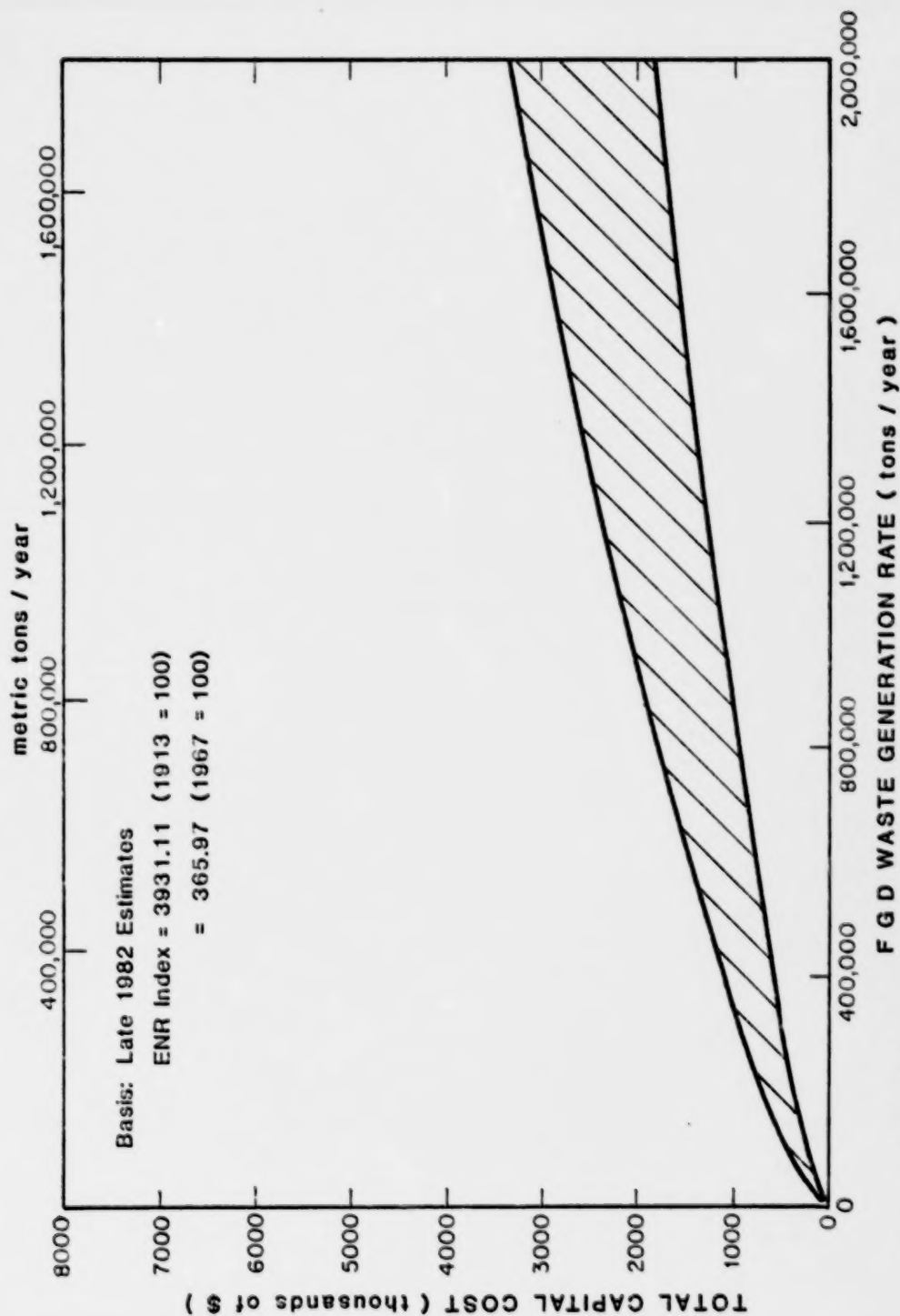


FIGURE 6.52 ANNUAL COST VERSUS BOTTOM ASH GENERATION RATE  
 BOTTOM ASH TRANSPORT ( Wet Sluicing )



Source: Arthur D. Little, Inc. Estimates

FIGURE 6.53 TOTAL CAPITAL COST VERSUS FGD WASTE GENERATION RATE  
 FGD WASTE TRANSPORT ( Wet Sluicing )

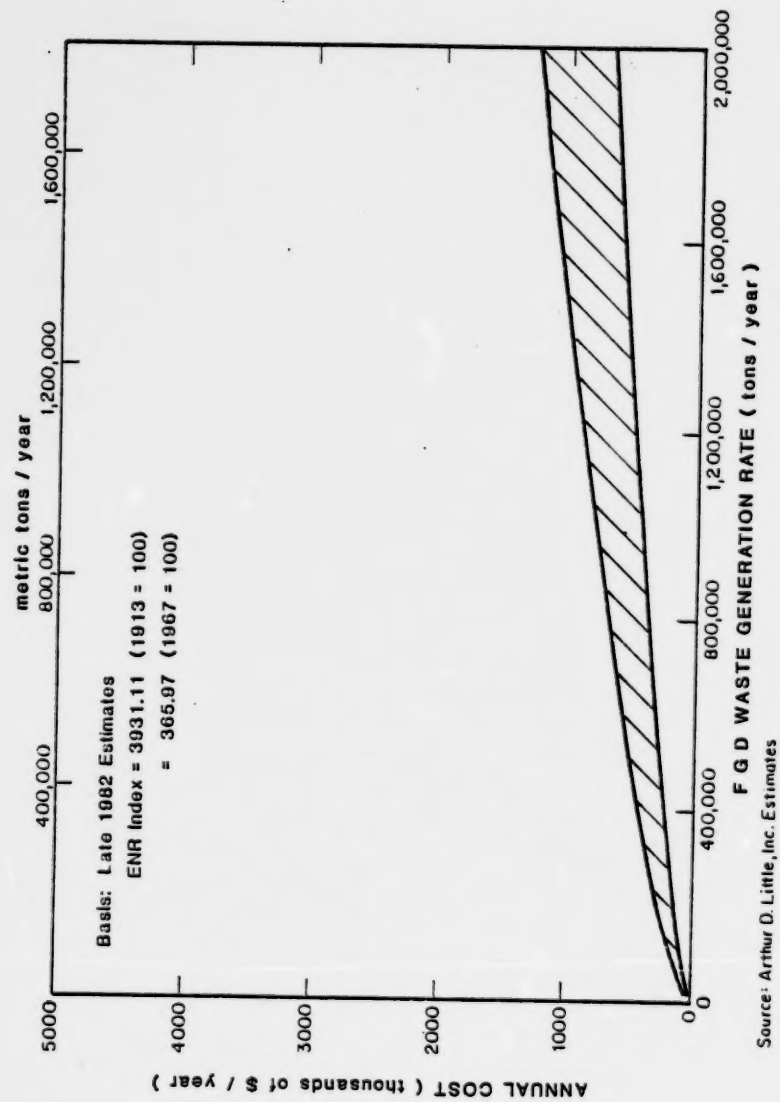
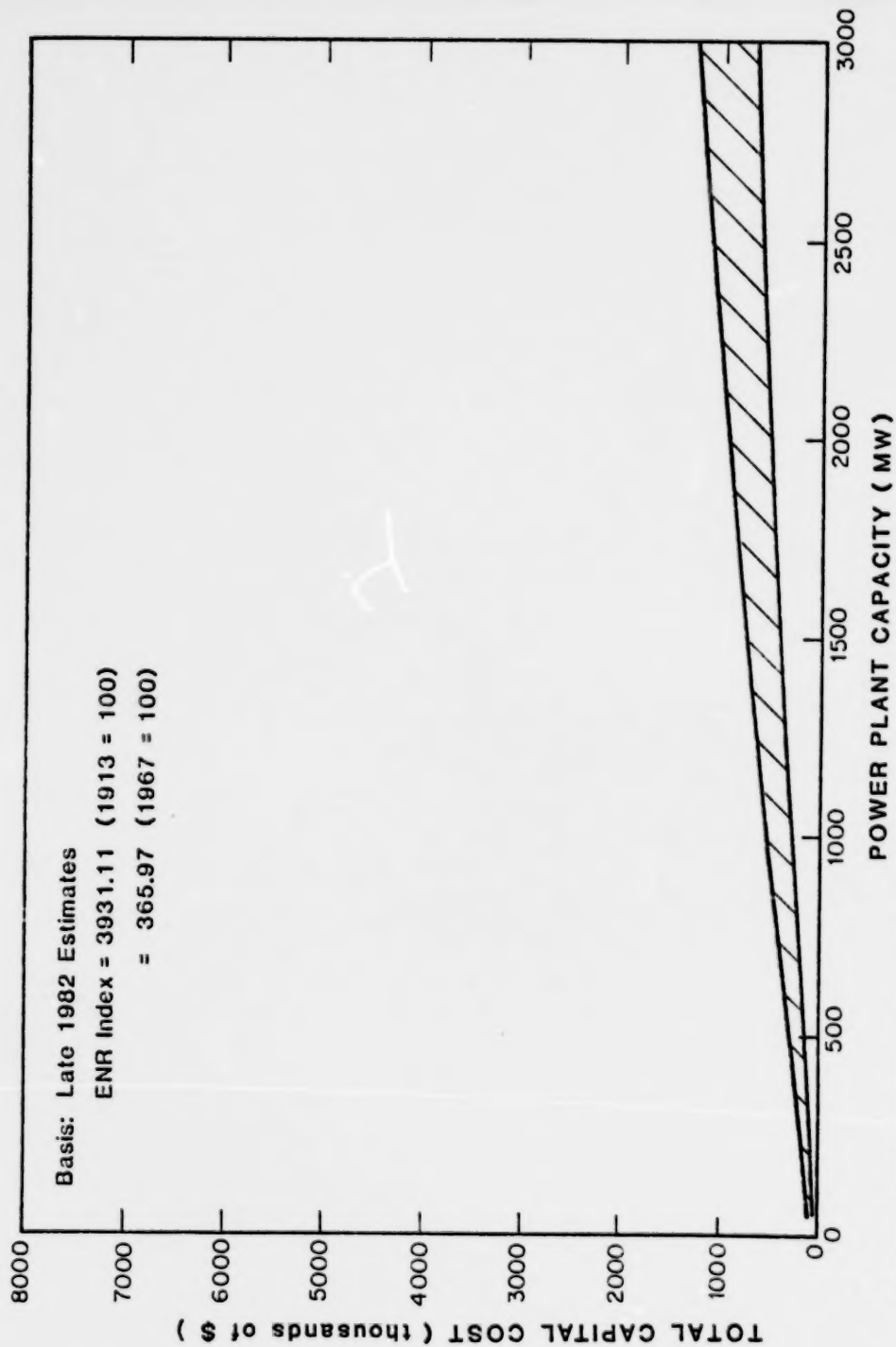


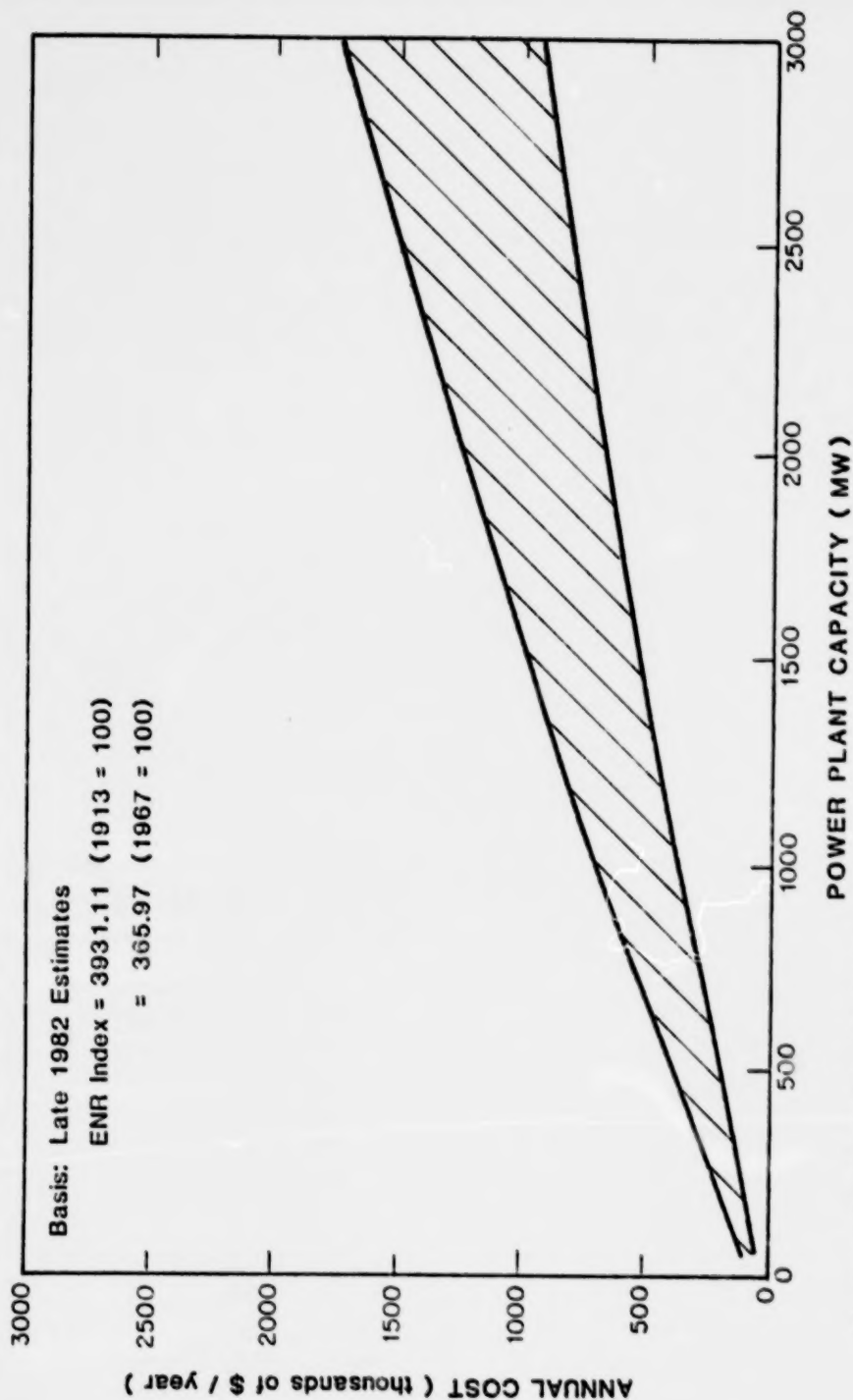
FIGURE 6.54 ANNUAL COST VERSUS FGD WASTE GENERATION RATE  
 FGD WASTE TRANSPORT ( Wet Sluicing )



Source: Arthur D. Little, Inc. Estimates

FIGURE 6.55 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
FLY ASH TRANSPORT (Trucking )





Source: Arthur D. Little, Inc. Estimates

FIGURE 6.56 ANNUAL COST VERSUS POWER PLANT CAPACITY  
 FLY ASH TRANSPORT (Trucking )

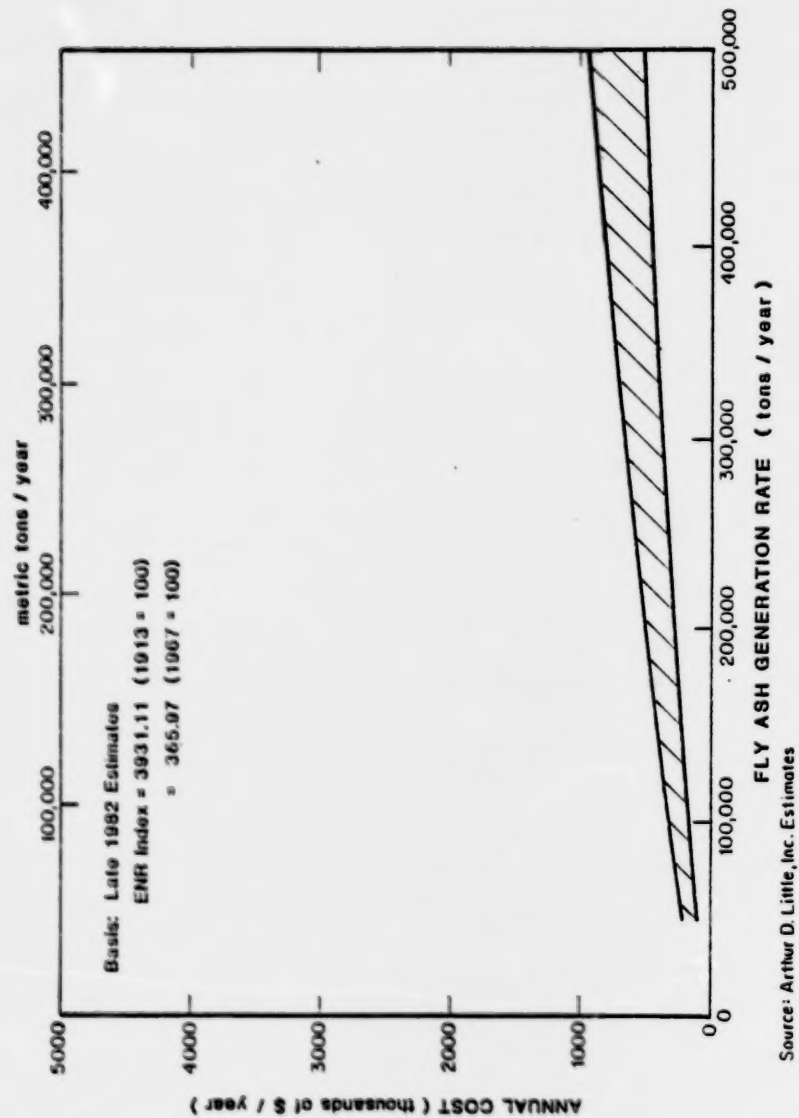
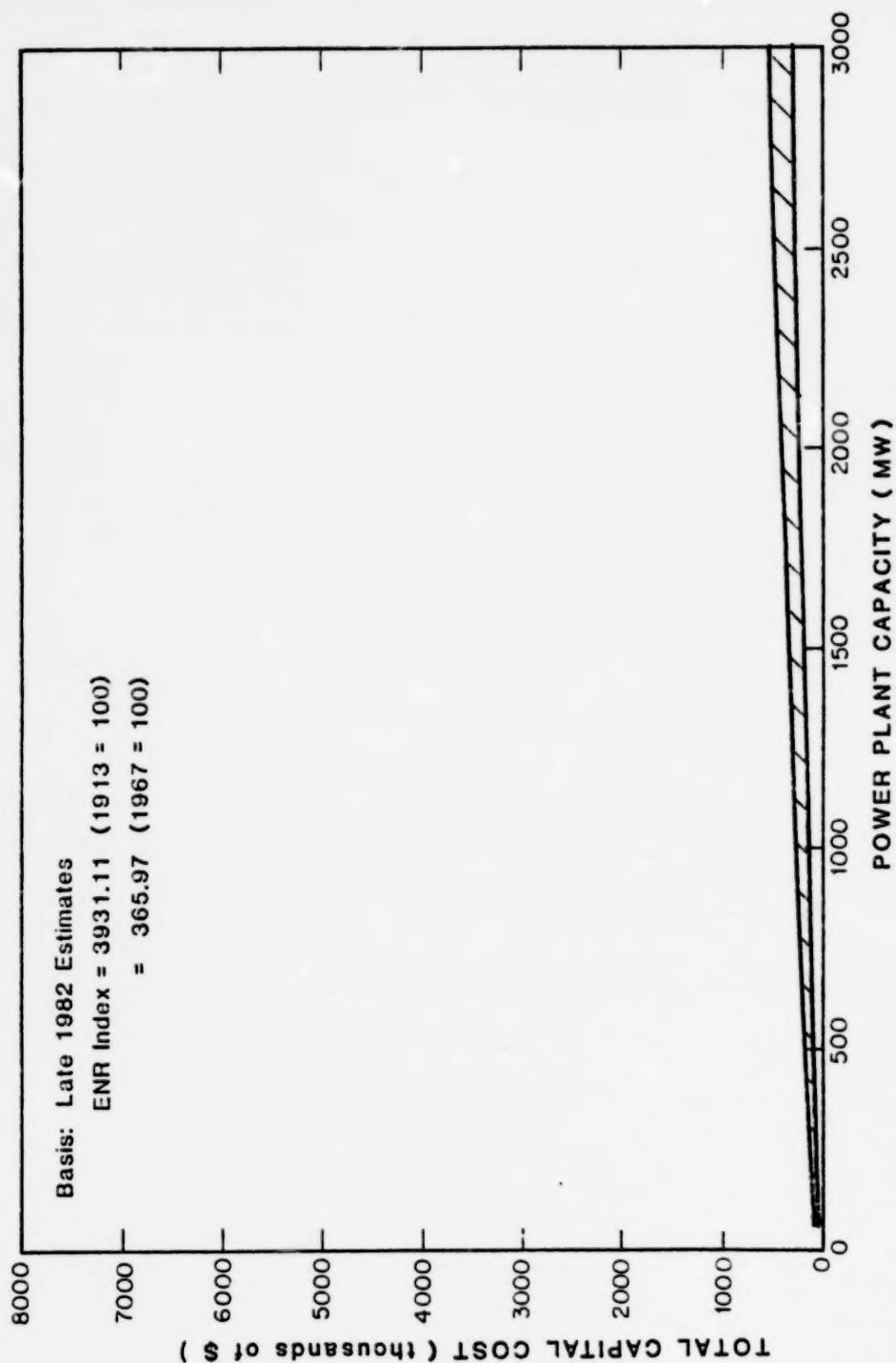
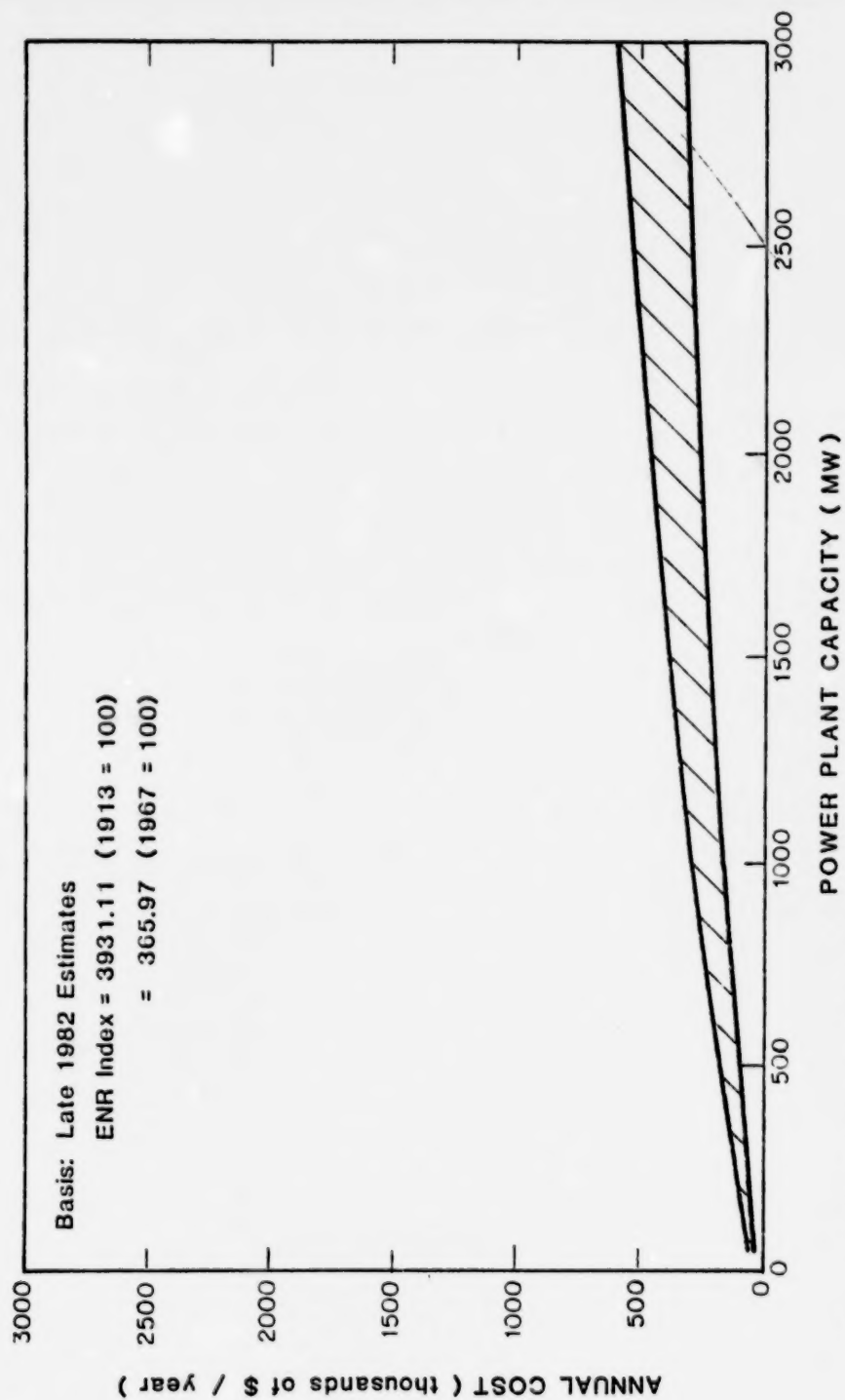


FIGURE 6.57 ANNUAL COST VERSUS FLY ASH GENERATION RATE  
 FLY ASH TRANSPORT (Trucking)



Source: Arthur D. Little, Inc. Estimates

FIGURE 6.58 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
 BOTTOM ASH TRANSPORT (Trucking)



Source: Arthur D. Little, Inc. Estimates

FIGURE 6.59 ANNUAL COST VERSUS POWER PLANT CAPACITY  
BOTTOM ASH TRANSPORT (Trucking)

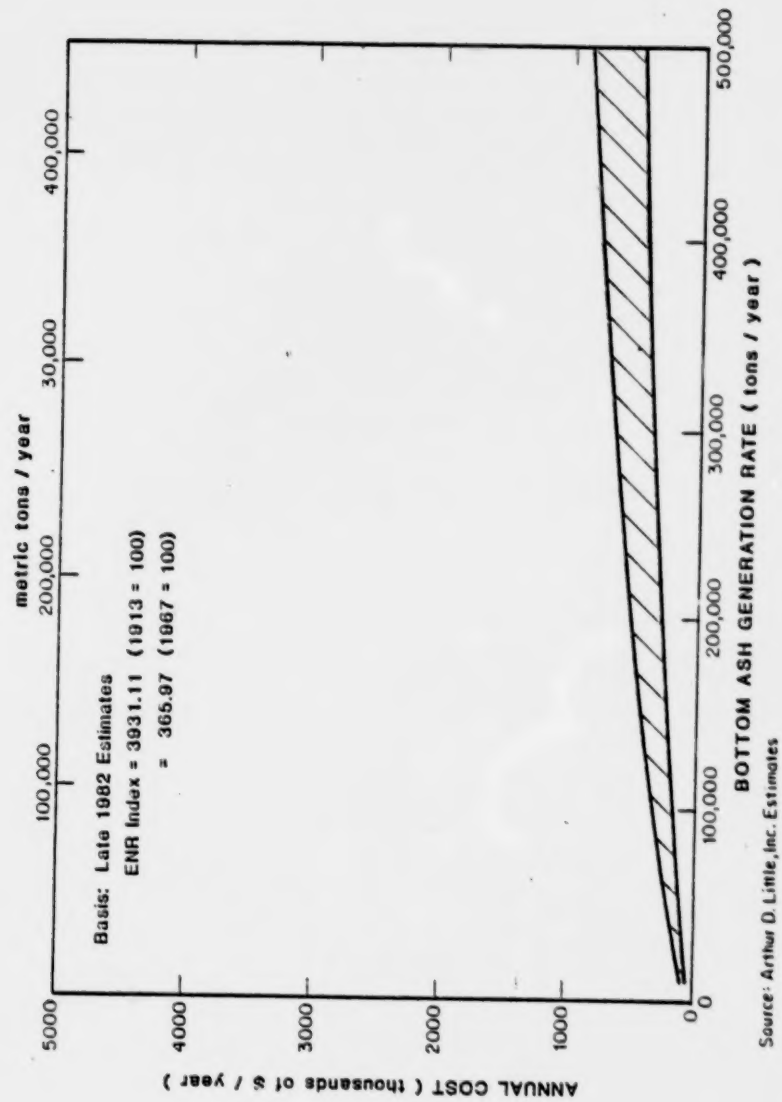


FIGURE 6.60 ANNUAL COST VERSUS BOTTOM ASH GENERATION RATE  
 BOTTOM ASH TRANSPORT (Trucking)

for annual costs) and FGD wastes (Figures 6.61 and 6.62 for capital and annual costs, respectively). Variations in these costs arise as a result of transport distance and transport vehicle type. For the purpose of constructing these cost curves, a transport distance of 1.6 km (1 mile) was assumed.

The effect of varying transport distance on truck transport costs was not determined (see Section 6.3). In general, increased transport distances require additional vehicles with accompanying purchase and operating cost increases. Such cost increases, however, would not be directly proportional to the amount of additional capacity required.

Coal-fired utility wastes can be transported in on-highway or off-the-road trucks. Off-the-road truck transport would result in higher capital but potentially lower annual costs. The curves are based on the use of on-highway trucks.

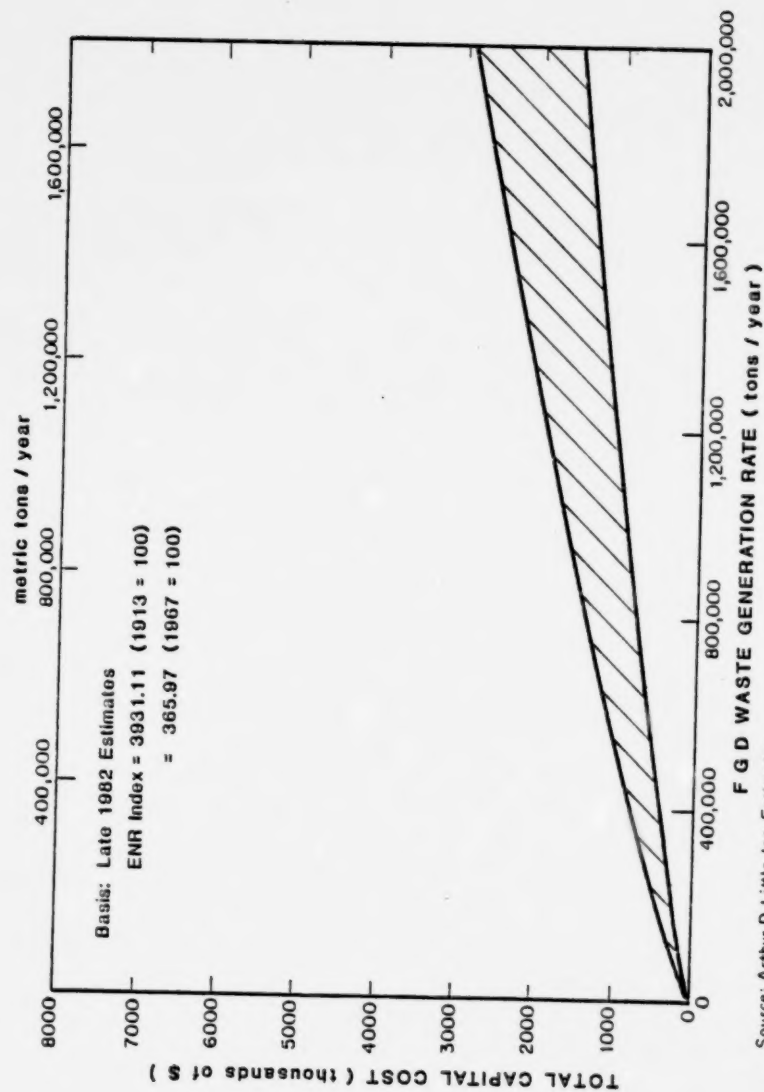
Coal Ash and FGD Waste Placement and Disposal -- There is very little difference among the three types of wastes with respect to disposal practices. In fact, codisposal of wastes is common practice. Thus, the following discussion of disposal capital and operating costs pertains to all of the three wastes under consideration.

Two types of disposal were considered with respect to capital and annual costs, ponding and landfilling. Mine disposal was not evaluated due to a paucity of data; ocean disposal, which is not currently practiced commercially, was similarly not considered.

Pond disposal costs for fly ash are provided in Figures 6.63 (capital costs), 6.64 and 6.65 (annual costs). Similar graphs are provided for bottom ash ponding, Figures 6.66 (capital costs), 6.67 and 6.68 (annual costs), as well as for pond disposal of FGD wastes, Figures 6.69 and 6.70 (capital and annual costs, respectively). Cost curves for landfill disposal are presented in Figures 6.71 (capital costs), 6.72 and 6.73 (annual costs) for fly ash; in Figures 6.74 (capital costs), 6.75 and 6.76 (annual costs) for bottom ash; and in Figures 6.77 and 6.78 (capital and annual costs, respectively) for FGD wastes.

Disposal site capital costs are primarily attributed to site preparation, excavation, and containment structure construction. With ponds, the costs of dike construction (for diked ponds) or excavation (for incised ponds) contribute significantly to overall capital costs. Sites which can be converted to ponds with minimal construction activities (e.g., valleys, abandoned pits, etc.) will have lower capital and annual costs than those which require significant earthwork. In the case of landfills, sites requiring minimal excavation to be suitable impoundment areas will have relatively lower costs than those requiring excessive earthwork. Thus, in disposal operations, the higher ranges of the capital cost curves reflect sites which require significant earthwork, while the lower cost band ranges pertain to disposal facilities where little earthwork was undertaken.





Source: Arthur D. Little, Inc. Estimates

FIGURE 6.61 TOTAL CAPITAL COST VERSUS FGD WASTE GENERATION RATE  
 FGD WASTE TRANSPORT ( Trucking )

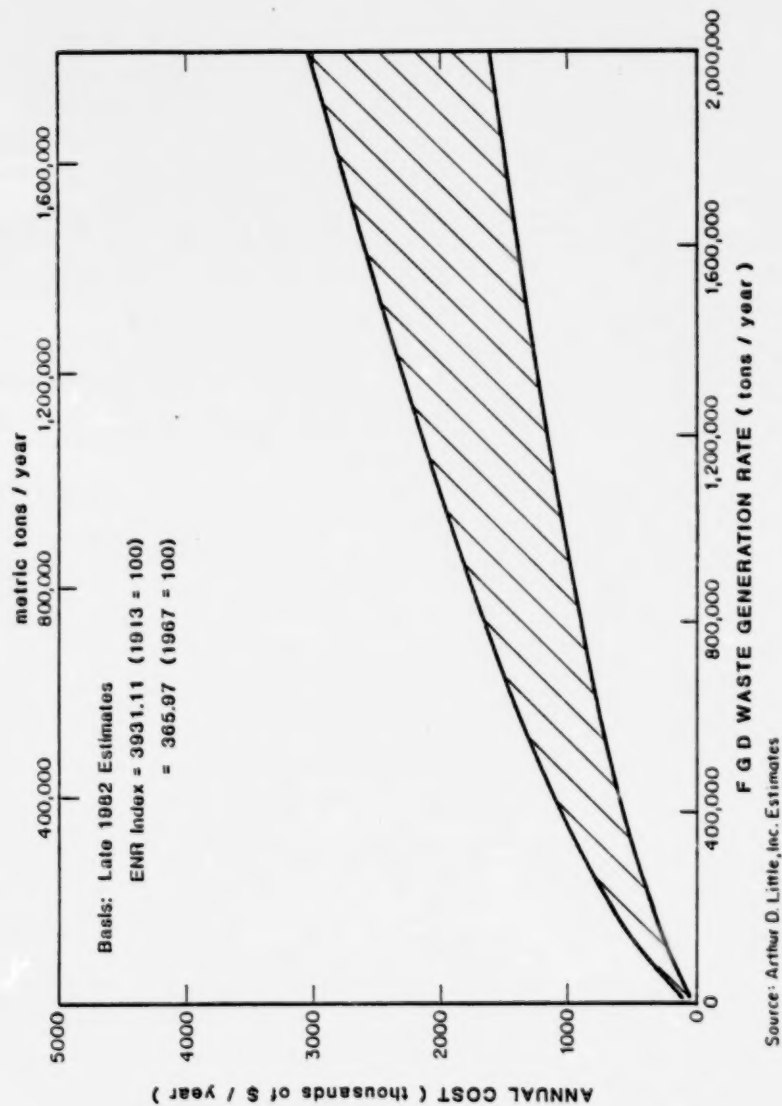
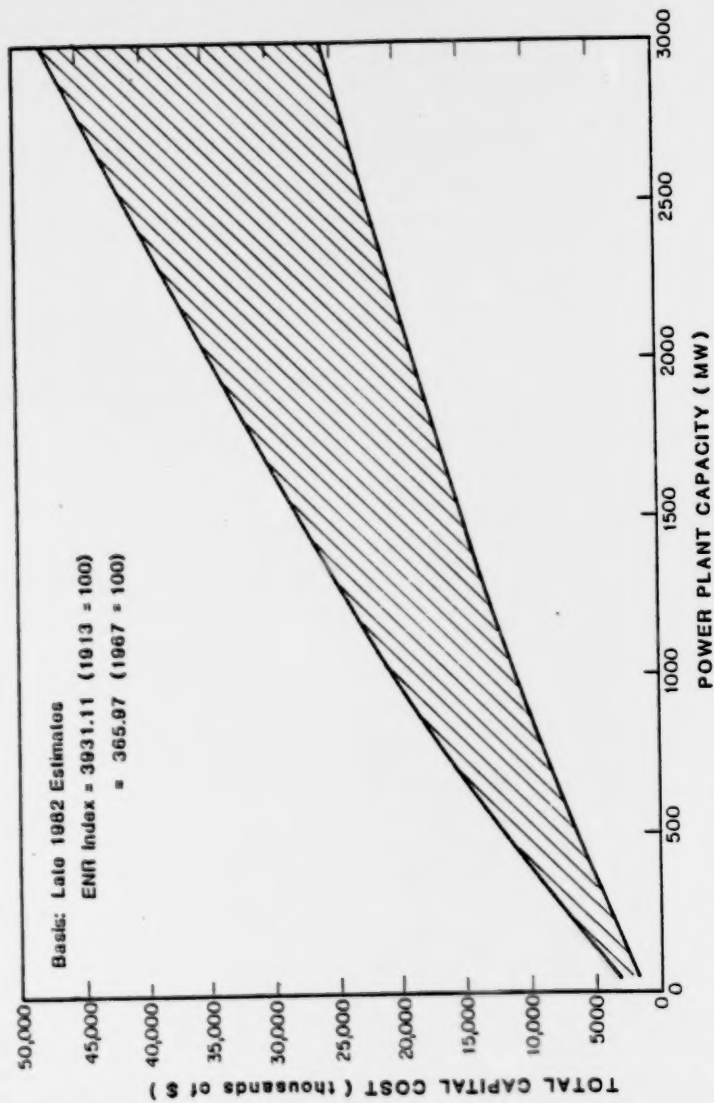
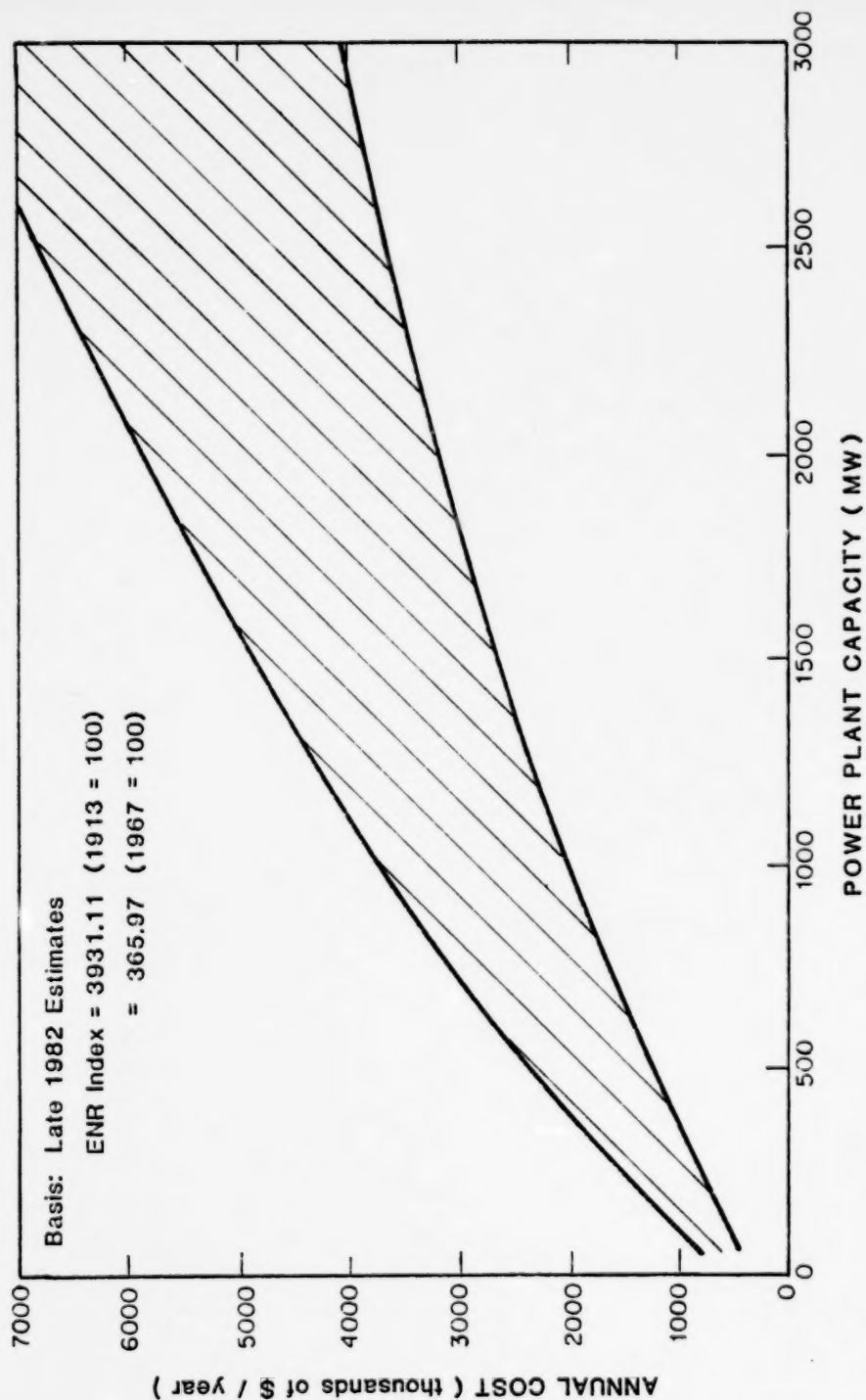


FIGURE 6.62 ANNUAL COST VERSUS F G D WASTE GENERATION RATE  
 F G D WASTE TRANSPORT ( Trucking )



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.63 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY**  
**FLY ASH PLACEMENT AND DISPOSAL**  
 ( Includes site monitoring and reclamation )  
 ( Ponding - Unlined Ponds )



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.64 ANNUAL COST VERSUS POWER PLANT CAPACITY**  
**FLY ASH PLACEMENT and DISPOSAL**  
 ( Includes site monitoring and reclamation )  
 ( Ponding - Unlined Ponds )

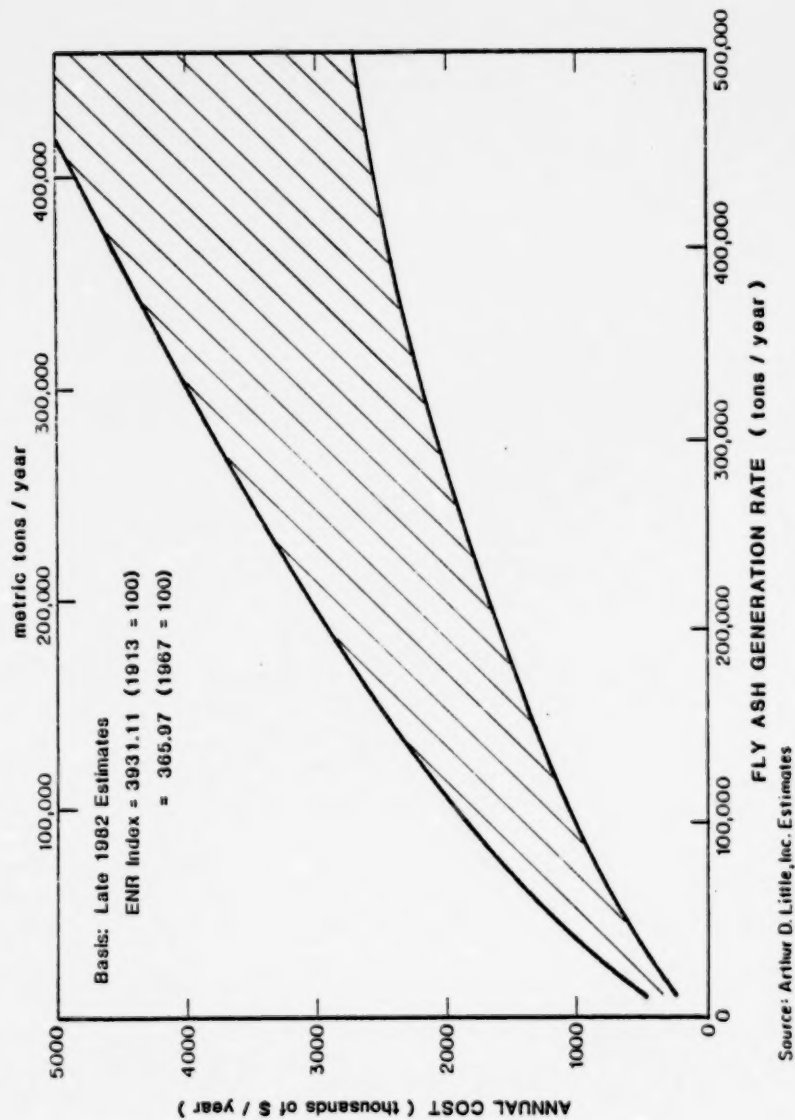
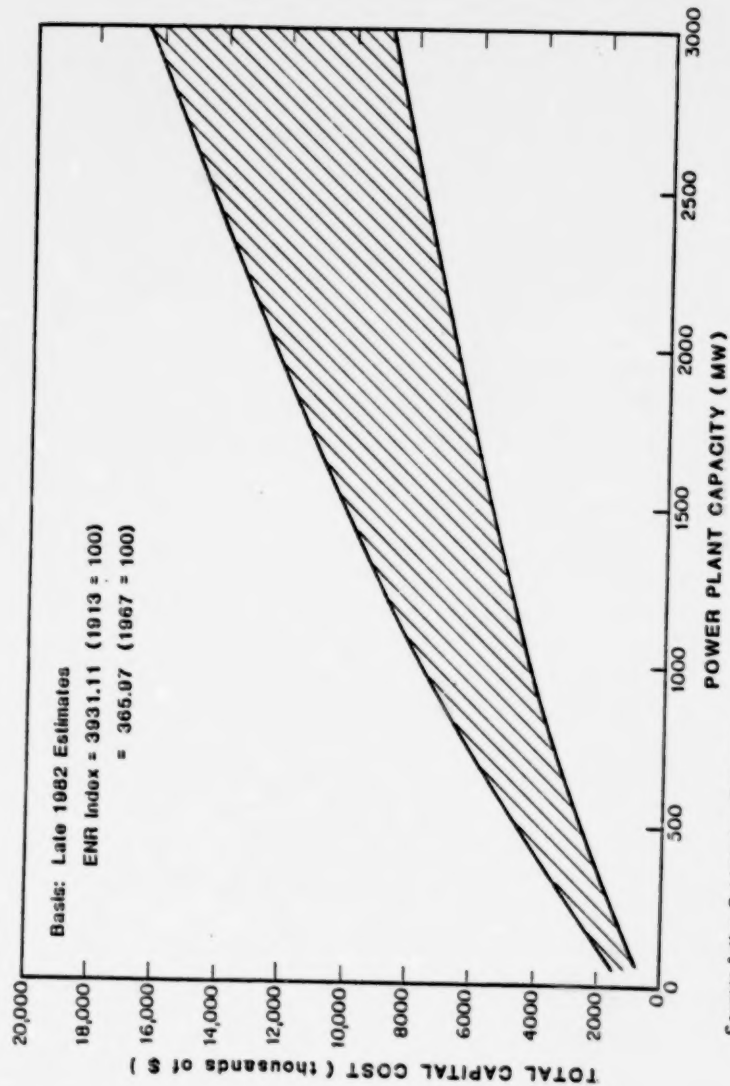


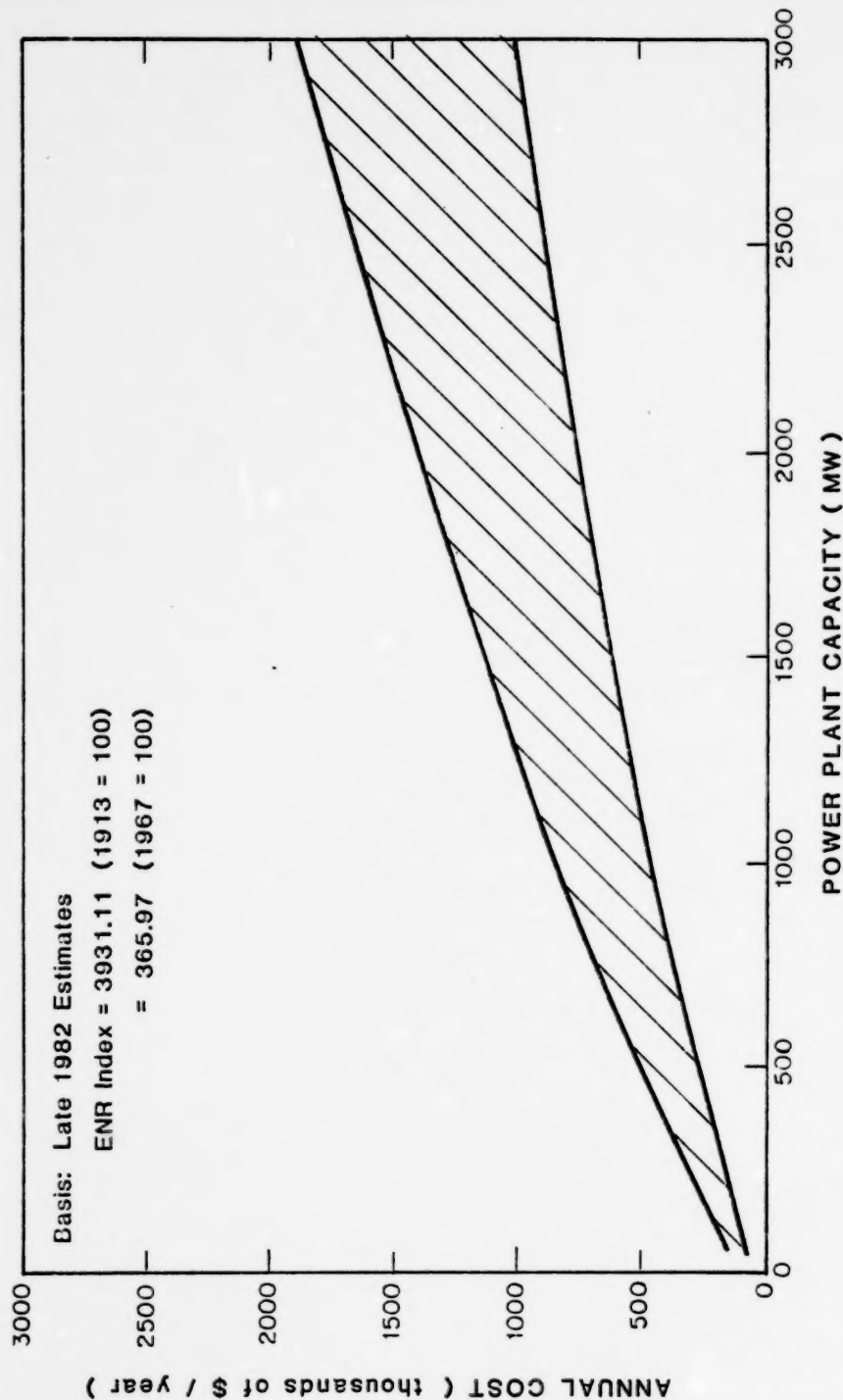
FIGURE 6.65 ANNUAL COST VERSUS FLY ASH GENERATION RATE  
 FLY ASH PLACEMENT AND DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Ponding - Unlined Ponds )



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.66 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
 BOTTOM ASH PLACEMENT AND DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Ponding - Unlined Ponds )**





Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.67 ANNUAL COST VERSUS POWER PLANT CAPACITY**  
**BOTTOM ASH PLACEMENT and DISPOSAL**  
 ( Includes site monitoring and reclamation )  
 ( Ponding - Unlined Ponds )

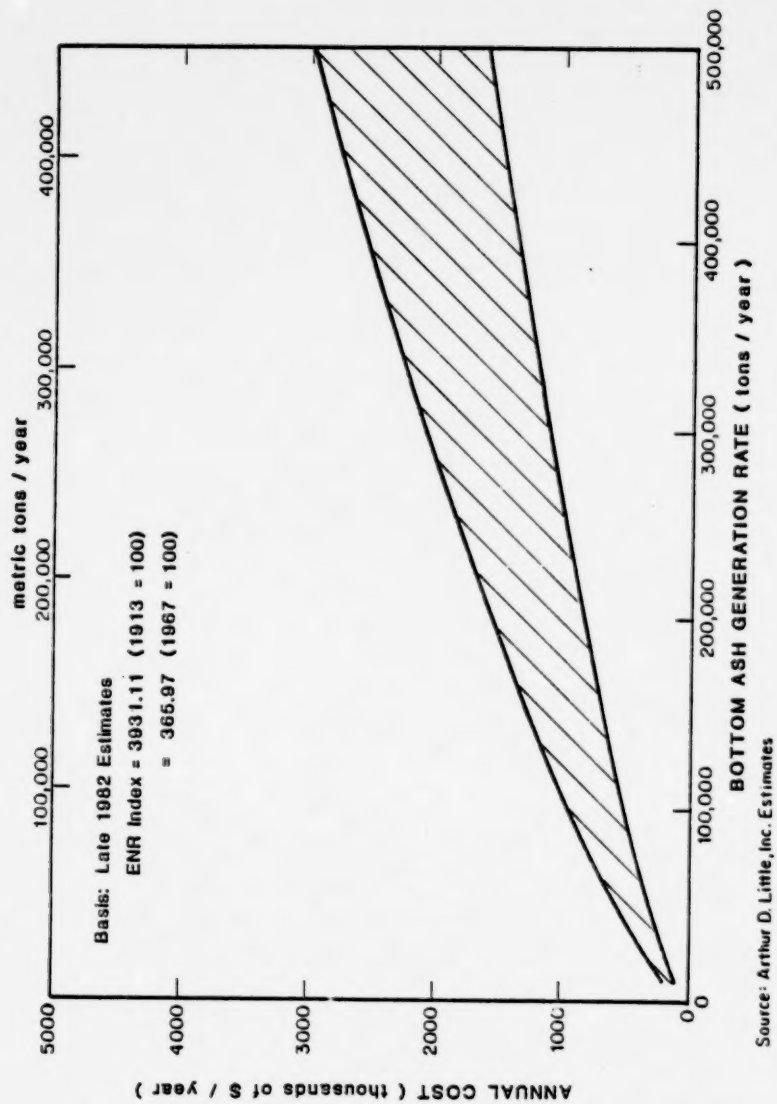


FIGURE 6.68 ANNUAL COST VERSUS BOTTOM ASH GENERATION RATE  
 BOTTOM ASH PLACEMENT AND DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Ponding - Unlined Ponds )

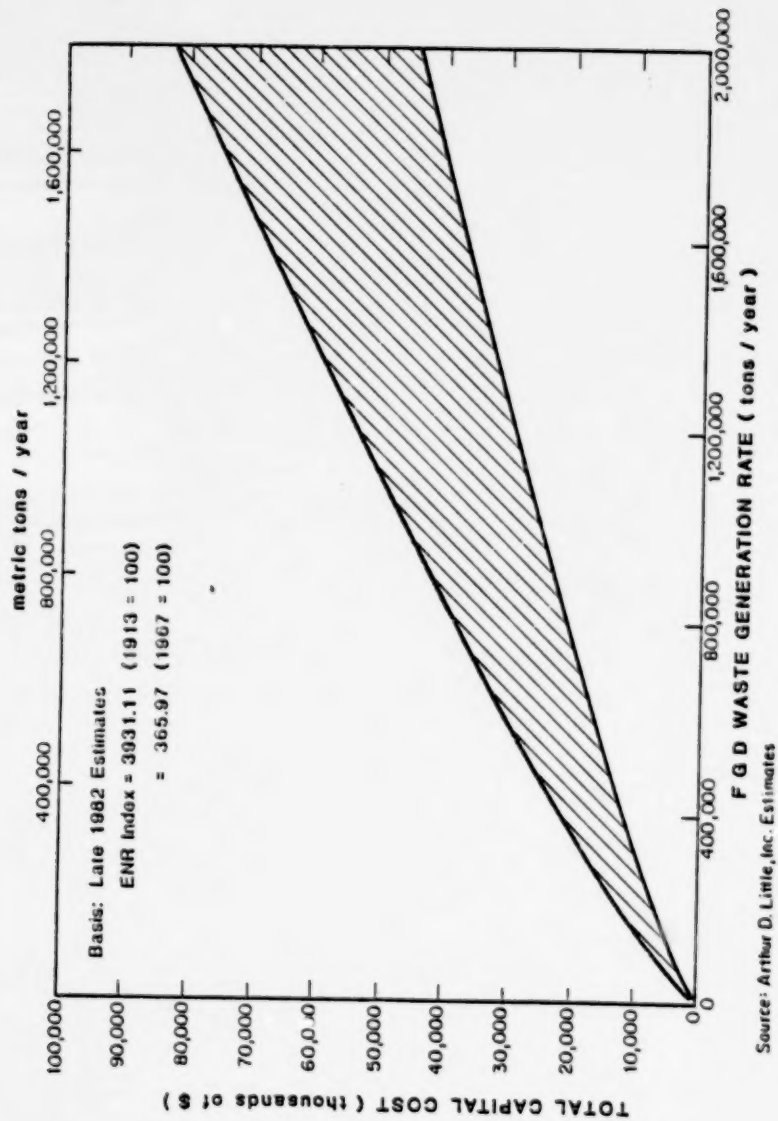


**FIGURE 6.69 TOTAL CAPITAL COST VERSUS FGD WASTE GENERATION RATE**  
**FGD WASTE PLACEMENT AND DISPOSAL**  
 ( Includes site monitoring and reclamation )  
 ( Ponding - Unlined Ponds )

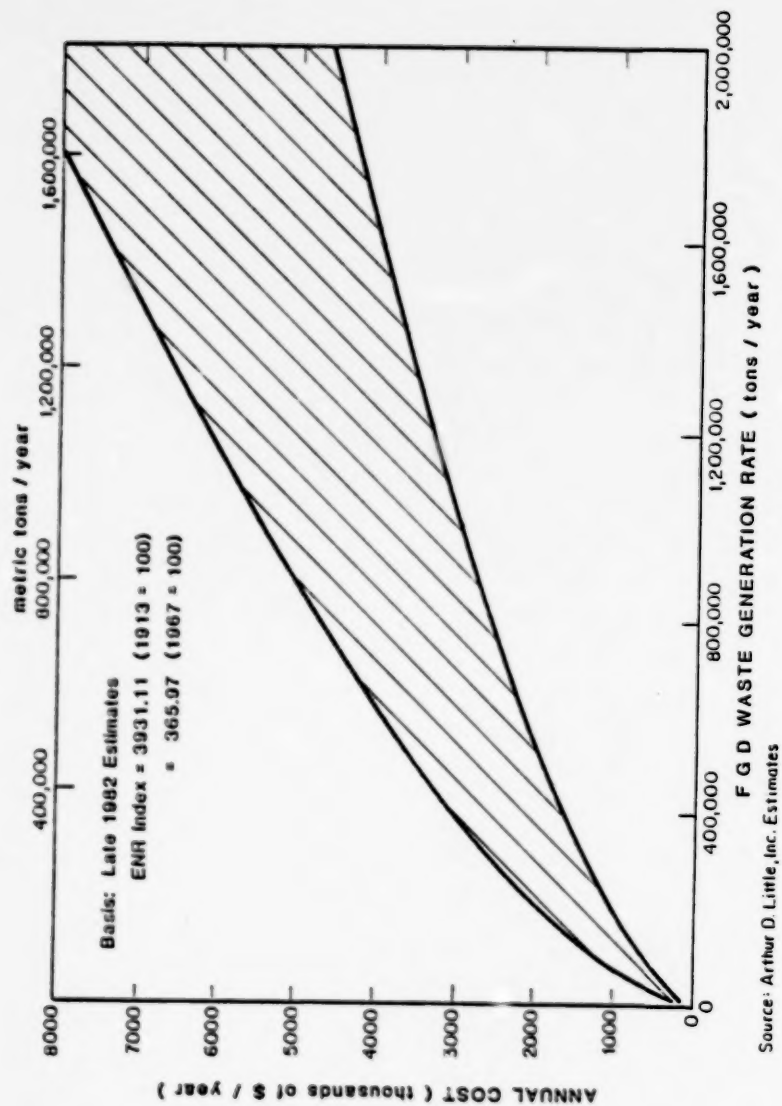


**FIGURE 6.70 ANNUAL COST VERSUS FGD WASTE GENERATION RATE**  
 FGD WASTE PLACEMENT AND DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Pounding - Unlined Ponds )

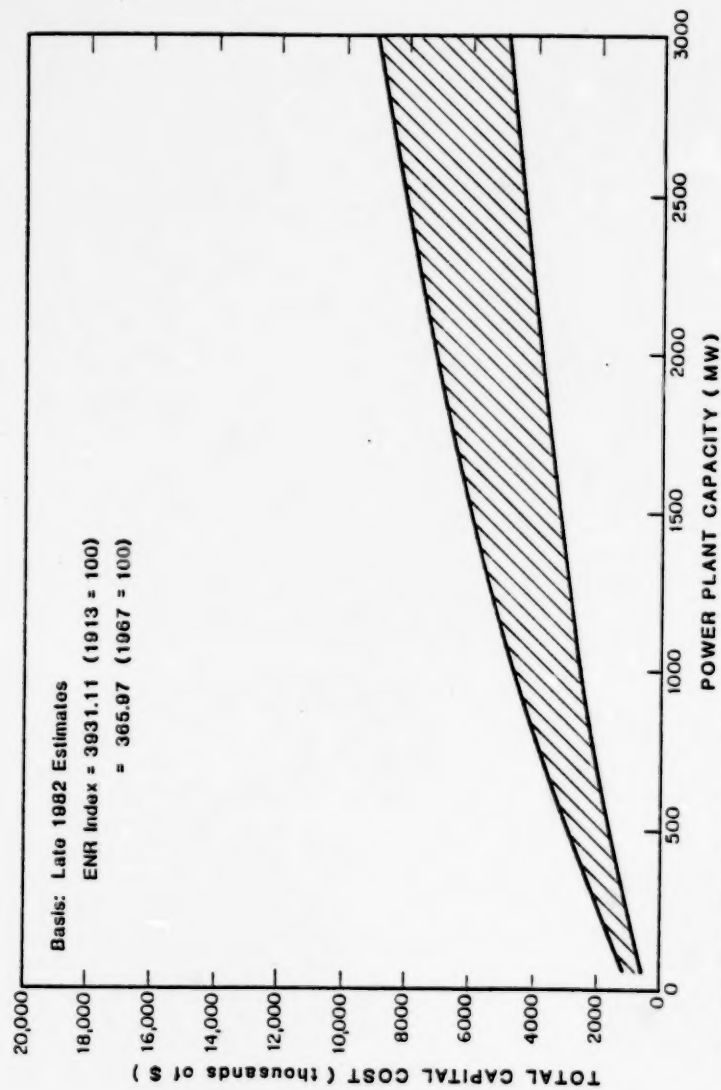
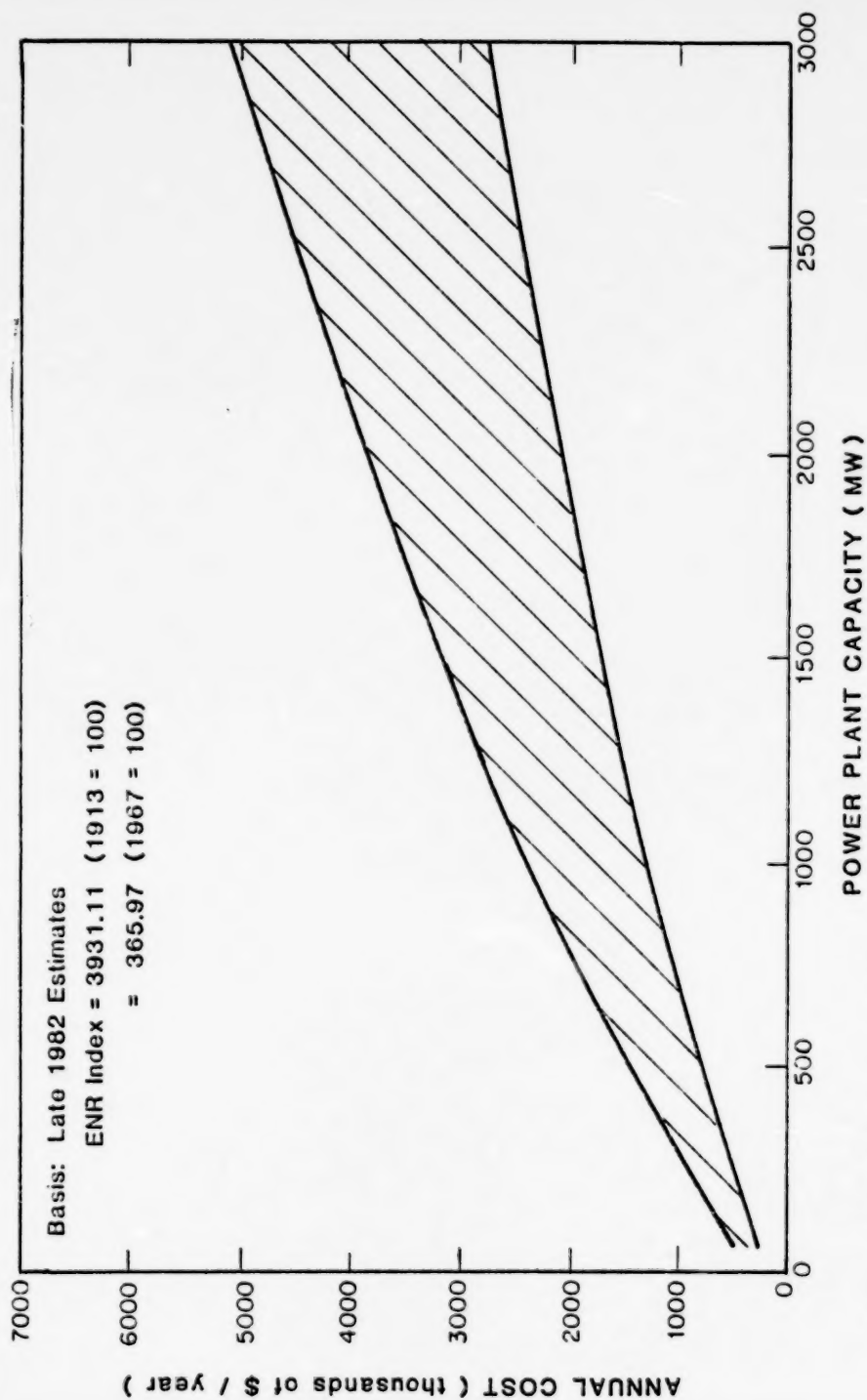


**FIGURE 6.71 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
 FLY ASH PLACEMENT AND DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Landfilling )**



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.72 ANNUAL COST VERSUS POWER PLANT CAPACITY**  
**FLY ASH PLACEMENT and DISPOSAL**  
 ( Includes site monitoring and reclamation )  
 ( Landfilling )



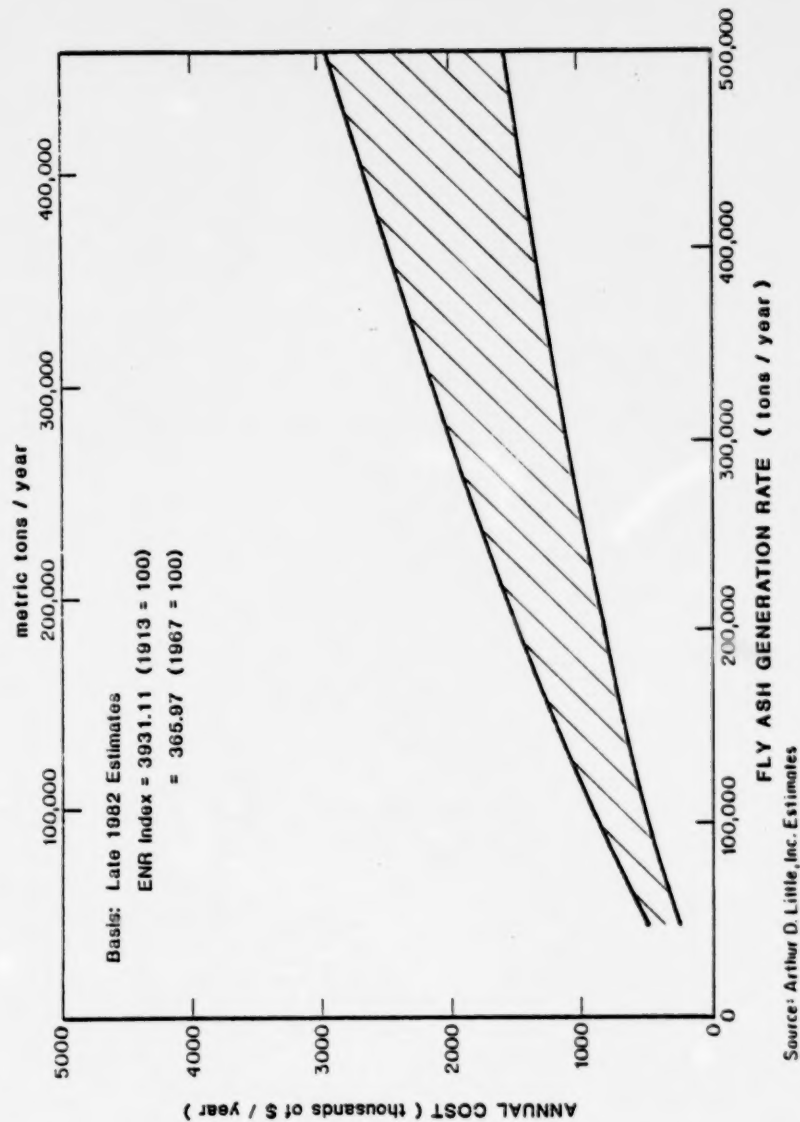
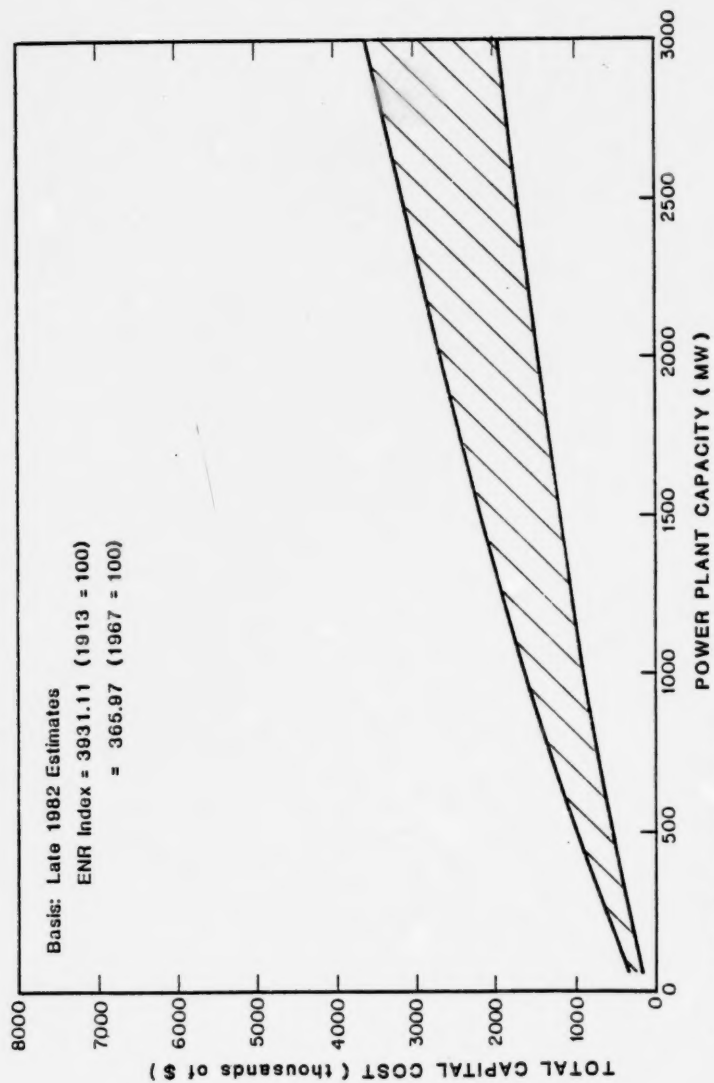


FIGURE 6.73 ANNUAL COST VERSUS FLY ASH GENERATION RATE  
 FLY ASH PLACEMENT AND DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Landfilling )



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.74 TOTAL CAPITAL COST VERSUS POWER PLANT CAPACITY  
 BOTTOM ASH PLACEMENT AND DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Landfilling )**

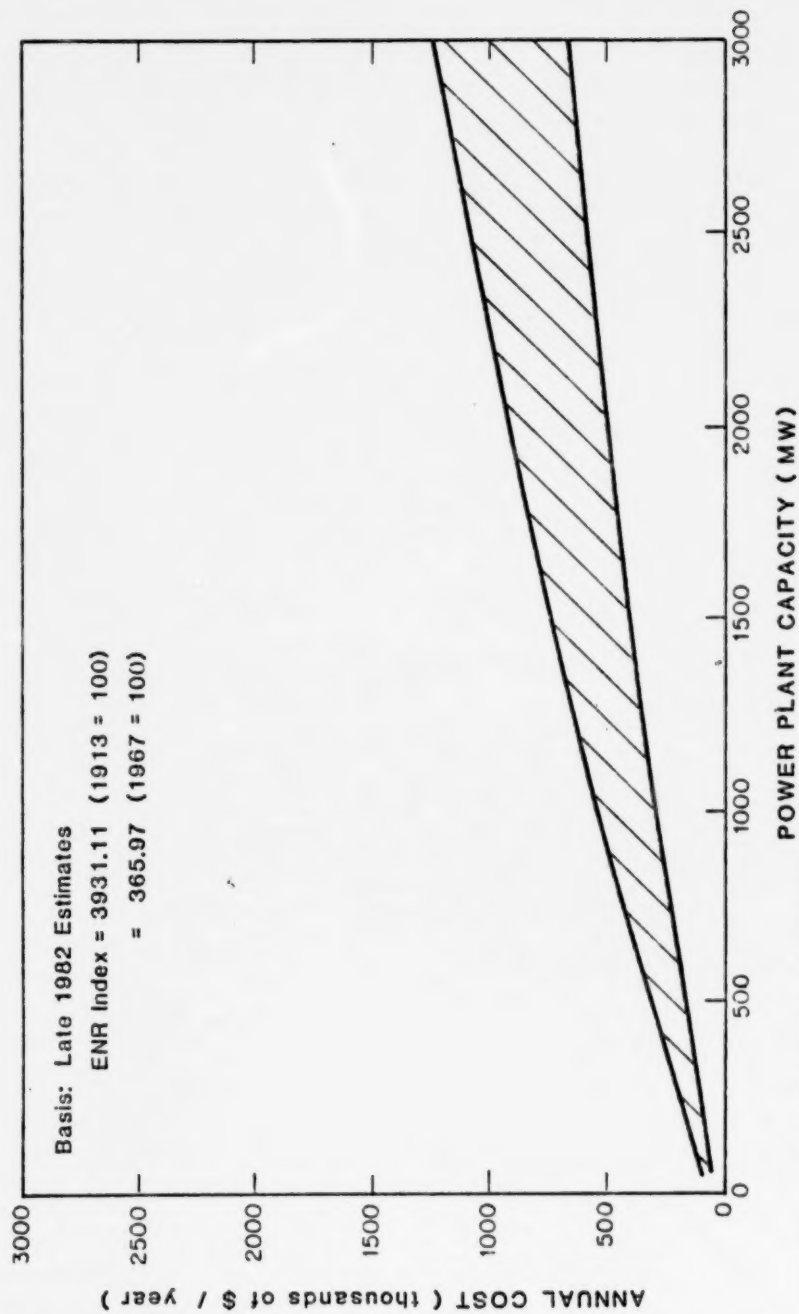


FIGURE 6.75 ANNUAL COST VERSUS POWER PLANT CAPACITY  
 BOTTOM ASH PLACEMENT AND DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Landfilling )

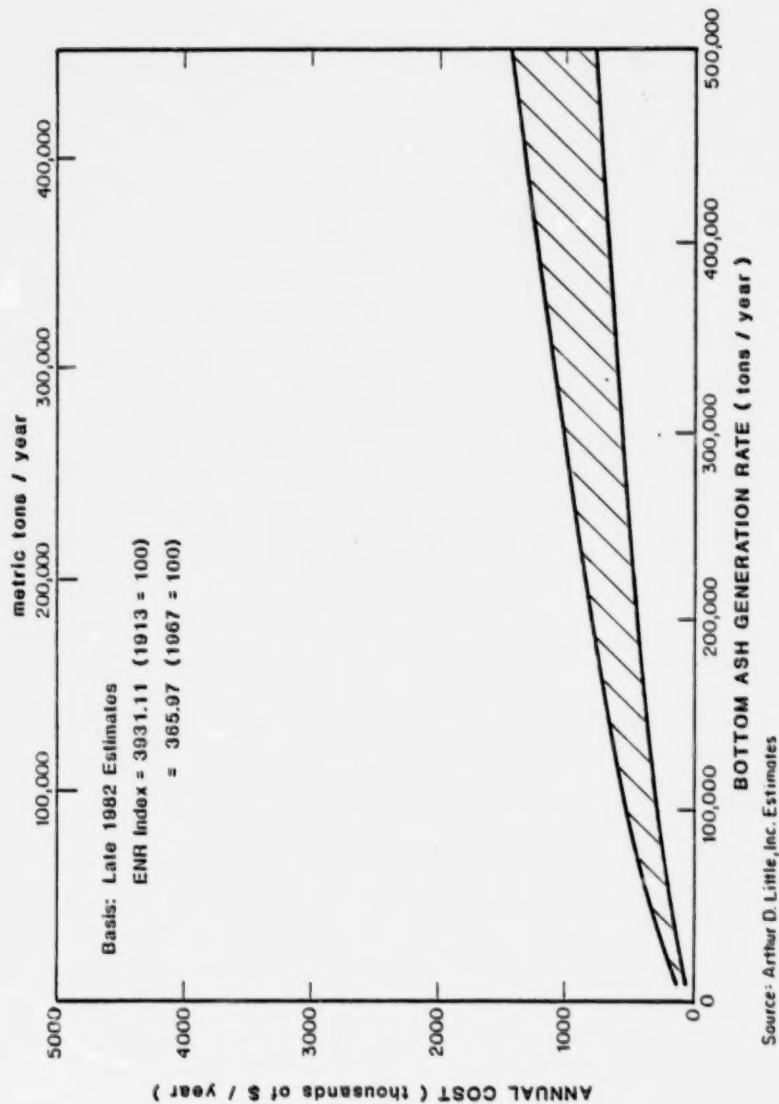
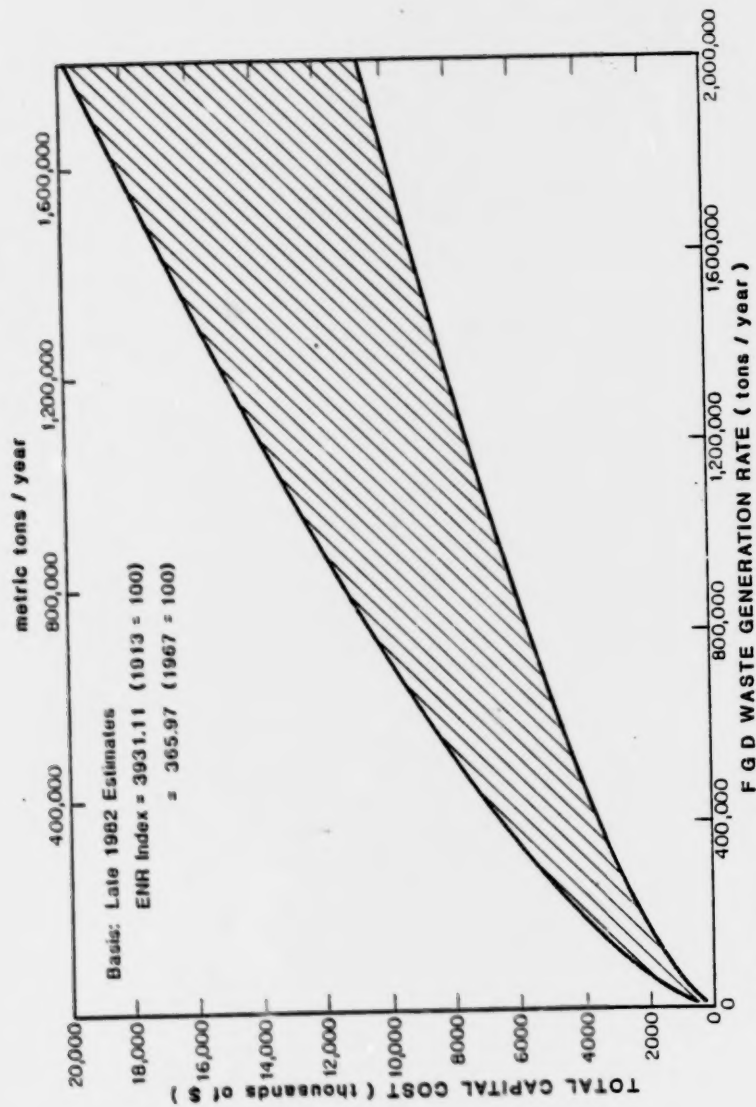


**FIGURE 6.76 ANNUAL COST VERSUS BOTTOM ASH GENERATION RATE**  
**BOTTOM ASH PLACEMENT AND DISPOSAL**  
 ( Includes site monitoring and reclamation )  
 ( Landfilling )



Source: Arthur D. Little, Inc. Estimates

**FIGURE 6.77 TOTAL CAPITAL COST VERSUS FGD WASTE GENERATION RATE  
FGD WASTE PLACEMENT AND DISPOSAL  
( Includes site monitoring and reclamation )  
( Landfilling )**

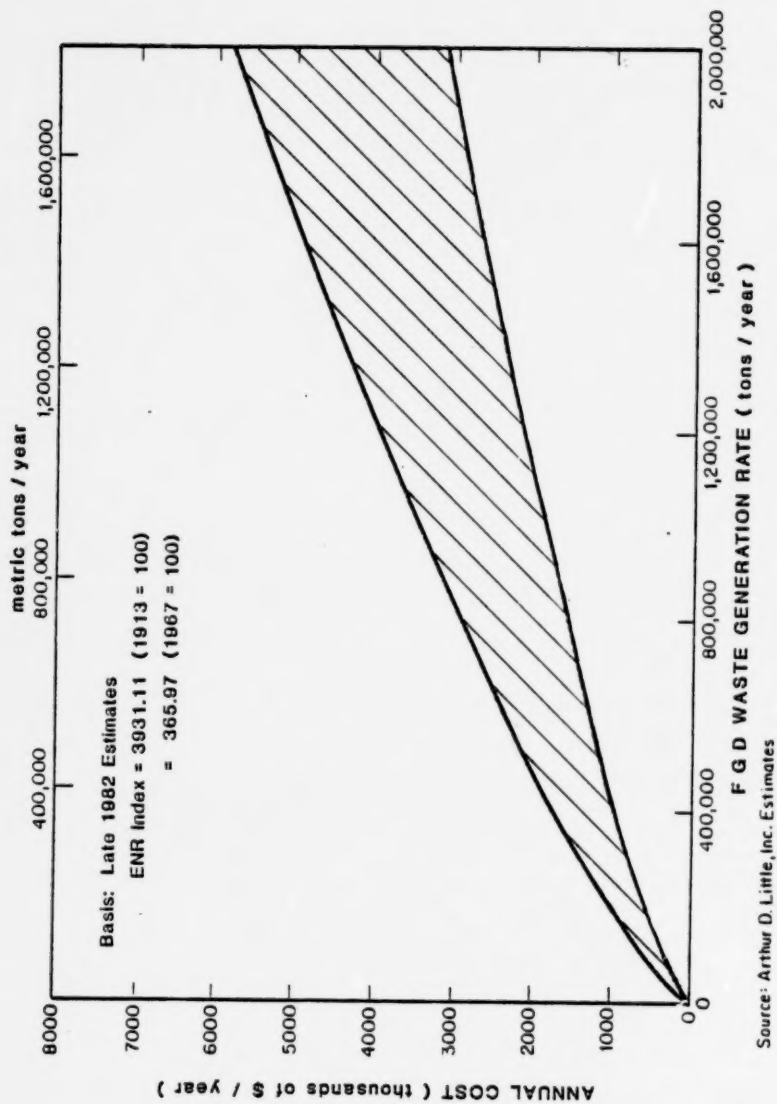


FIGURE 6.78 ANNUAL COST VERSUS FGD WASTE GENERATION RATE  
 FGD WASTE PLACEMENT and DISPOSAL  
 ( Includes site monitoring and reclamation )  
 ( Landfilling )



Another issue with respect to landfill disposal is posed by operating requirements. In some cases, wastes are dumped with moderate or minimal compaction, while in other cases, significant effort is expended to place the waste in a specific manner. For example, the wastes from CSI chemical fixation processes are routinely compacted and graded at the landfill site. Thus, landfill annual costs are highly dependent upon placement requirements as well as on the capital charges contribution. In cases where the landfill requires significant construction and placement of the waste involves compaction, the annual costs will fall in the upper ranges of the cost band. Where no significant capital charges or compaction requirements are involved, annual costs will be much lower.

The most significant annual cost for pond disposal is for capital charges. Thus, pond configurations which give rise to high capital costs (i.e., those which require significant site preparation and earthwork) would be likely to have high annual costs.

All disposal cost curves pertain to sites which do not have natural (i.e., clay) or synthetic liners. Liner costs are highly site specific depending on:

- the liner material employed;
- the availability of a good source of liner material within a reasonable proximity, in the case of natural liners;
- the desired thickness of the liner; and
- the area to be lined.

Due to this site-specificity, and the fact that a very large number of design options exist for disposal site liners, it was not possible to develop generic cost curves which adequately included all such site conditions and options. Instead, some general insights into liner costs and factors influencing variations in such costs are presented below.

A variety of lining materials are available for disposal operations. Clays, polymeric membranes, admixtures, chemical sealants, and other materials have been used. (A more detailed discussion of liner characteristics is presented in Section 6.1.6.2) The most common liners for utility waste disposal applications include clays, polymeric materials and stabilized utility wastes. This study considers only those three liner types.

Clay Liner materials include a variety of clay soils. However, the most well-known clay liner material is bentonite. The most important cost consideration with respect to clay liners is the availability of the liner material; transportation is a major cost element if the clay must be imported. Compacted indigenous clays are low-cost liner options. Permeability can vary, depending on the nature of the clay and degree of

compaction. Thus, proper installation is essential to optimum performance. Imported bentonite is higher in cost than native soil liners, although the use of this liner may, in some cases, be more economical than polymeric liner options. Of course, this is a site-specific issue relating to the cost of transporting the bentonite from its source to the disposal site. The installed cost (1982 dollars) of clay liners ranges from \$5.10 to \$18.00/m<sup>3</sup> (\$3.90 to \$13.80/yd<sup>3</sup>) (59). Assuming a liner thickness of 0.46 m (18 in), this corresponds to a cost range of \$2.30 to \$8.20/m<sup>2</sup> (\$9,400 to \$33,000/acre). Transportation of imported clays, if such is necessary, can influence cost trends significantly. Assuming a bentonite source in Wyoming, transportation costs to the West Coast, East Coast and the South could be in the ranges of \$55 to 65/metric ton (\$50 to 65/ton), \$130 to 140/metric ton (\$120 to 130/ton) and \$105 to 110/metric ton (\$95 to 100/ton) for each destination, respectively (60). The cited costs were adjusted to late-1982 dollars. The clay liner costs presented here are for placement of the liner only and do not include site preparation costs (e.g., clearing, grubbing, cutting, filling, etc.) which are attributed to disposal site construction. The liner costs are presented in late-1982 dollars and include the total incremental capital investment for liner placement. Items such as engineering and contractor's overhead and fees, as well as contingency, are included.

Stabilized utility waste materials have been used as liner materials. One material, Poz-O-Tec®, is a mixture of fly ash, FGD waste and lime; a second material which is a mixture of only coal ash (fly ash and slag) and lime is currently denoted Pozzolanic Aggregate Mixture (PAM), although it was previously referred to as Poz-O-Pac®.

The fixated waste may be an attractive liner for new disposal sites at existing utility plants, since the majority of the liner material is available captively as an undesired by-product. For new plants, it may be necessary to import the liner materials, at least during the initial phases of plant construction.

Based on discussions with a supplier of Poz-O-Pac® liners, a fully installed cost of \$15.70/m<sup>3</sup> (\$12.00/yd<sup>3</sup>) was estimated (M. Nowicki, American Fly Ash Company, personal communication, September 24, 1982). For hypothetical sites with 0.9 m (3 ft) and 1.5 m (5 ft) liner thicknesses, this corresponds to a cost of \$14.40/m<sup>2</sup> (\$58,000/acre) and \$24.00/m<sup>2</sup> (\$97,000/acre), respectively. The costs cited here were estimated with the underlying assumption that all required materials are locally available.

Polymeric membranes have been increasingly applied as waste disposal site liner materials. Two types of polymeric membranes are in use, exposable (i.e., membranes which are resistant to exposure to the elements without being covered) and unexposable (i.e., membranes which do not retain their integrity if left exposed). Of the four common polymeric membrane liner materials, three fall into the exposable category (i.e., hypalon, chlorinated polyethylene, and high density polyethylene), while one, polyvinyl chloride (PVC) is an unexposable material. The installed costs

(late-1982 dollars) for the listed types of exposable membrane liners range from \$9.40 to \$24.50/m<sup>2</sup> (\$38,000 to \$99,000/acre), and those for unexposable membranes range from \$12.80 to \$26.70/m<sup>2</sup> (\$52,000 to \$108,000/acre) (60). The above costs include soil sterilization, subgrade preparation, underliner materials and installation, liner materials and installation, contractor's profit and overhead, engineering fees/overhead and contingency. The ranges provided reflect differences in the cost of varying materials, differences in cost related to liner thickness, and a contribution for minimal site specific requirements. The cost of site preparation (e.g., clearing, grubbing, filling and compaction) is not included, since these have been attributed to disposal site construction. As such, the costs presented are incremental costs for liner placement only.

The costs associated with site monitoring and reclamation are included in the waste placement and disposal module costs. Site monitoring capital costs, based on the assumption that six groundwater monitoring wells are installed for any one disposal site, would typically range in the vicinity of \$15,000 to 20,000. As one would expect, site monitoring costs contribute very little to the overall waste placement and disposal module capital costs. The same holds true for the annual cost associated with site monitoring (sampling and analysis) activities, which generally fall in the range of \$8,500 to 12,000/year for the first year (assuming four sampling trips per year). In contrast, site reclamation costs are highly site-specific, depending on the surface area to be reclaimed, the availability of top soil, and the need for dewatering of the waste in the case of ponding operations. Site reclamation activities covered by the waste placement and disposal module costs in this project consist of covering the disposal site with 0.45 m (1.5 ft) of cover soil along with 0.15 m (6 in) of top soil (not available on site), followed by seeding. The capital cost for such reclamation would typically be on the order of \$1.50 to 2.20/m<sup>2</sup> (\$6000 to 9000/acre).

Related Environmental Systems (Air Pollution Control) -- As noted in Section 6.1.2, in developing engineering and cost data on solid waste handling and disposal systems at utilities, it is difficult to divorce oneself entirely from other aspects of a plant's environmental control system. One such example is the air pollution control system, which consists of particulate control, and in some cases sulfur oxides control. Using broad capital and annual cost estimates for both the air pollution control and the waste handling and disposal systems, one can better understand the cost contributions of each to the total environmental control system cost.

The cost of air pollution control systems sheds some light on the cost of coal-fired utility waste disposal by allowing for the assessment of integrated environmental control systems. However, detailed capital and operating cost estimates for air pollution control were not developed in this project. The focus was rather on developing broad unit cost data of particular usefulness for preliminary evaluation of waste disposal options.

Presently, two types of air pollution control systems are used by coal-fired utilities, sulfur dioxide control systems and particulate control systems. A variety of options are available with respect to each type of air

pollution control system. The cost data in this study are limited to include only those systems most commonly employed by utilities. Thus, with respect to sulfur dioxide control, conventional lime and limestone scrubbing systems are considered. Electrostatic precipitation is considered with regard to particulate control.

The cost of sulfur dioxide emissions control is affected by a number of variables, including, but not limited to:

- Plant size - FGD systems are modular in nature. One would expect costs per module to decrease as more identical modules are used. Hence, FGD costs per unit electric power generating capacity are expected to increase with decreasing plant size. In other words, economies of scale can be realized with FGD systems.
- Coal sulfur content - For specific sulfur oxides emission control requirements, the use of coal with relatively lower sulfur contents would result in the need for relatively lower sulfur oxides scrubbing capacities. Lower capacities, likewise, are lower in cost.
- Reagent consumption and availability - There are certain trade-offs between reagent availability and stoichiometric requirements. However, in some cases the less efficient reagent may be the least costly, particularly when a more efficient reagent is not available locally at an economical price.

In light of these variables, a review of the literature indicates that capital costs (in late-1982 dollars) for new FGD installations, typically lime or limestone based, would range from \$90 to 170/kW (61-63). For retrofit applications, the capital costs for similar FGD systems could be up to 30 percent higher. Operating costs for these same systems would range from \$35 to 65/kW per year. It must be noted, however, that a wide variability exists among the reported costs in the literature. This is primarily because the capital and operating costs found in the literature, as presented here, are attempting to represent a range of plant sizes (200 to 2000 MW), coal sulfur contents, and reagent consumption and availability. In addition, the wide variability in reported costs from different sources stems from plant specific conditions, including:

- design requirements for meeting various emission standards;
- battery limits included in the engineering/cost estimate; and
- degree of design redundancy in the FGD system itself.

The capital cost range noted above for these FGD systems covers direct and indirect capital costs. Direct capital costs include purchased equipment costs and installation costs (materials and labor) for system battery limits including the following major process areas: raw material handling and storage; feed preparation; flue gas treatment; and flue gas reheat. Waste handling, processing and disposal are not included in these estimates.



However, indirect capital costs such as field supervision, construction and field expense, and engineering and contingency are included. The annual operating costs similarly include both direct and indirect annual costs. Direct annual costs are for such items as operating labor, materials (e.g., lime or limestone), supervision, maintenance (materials and labor) and utilities (e.g., power and water). Indirect costs would include such elements as overhead expenses (i.e., payroll and plant overhead), general and administration costs, taxes, insurance, and capital charges.

As with FGD systems, the cost of particulate control by electrostatic precipitation is affected by a number of variables, including:

- electrical resistivity of the fly ash;
- temperature of the flue gas; and
- flue gas composition.

Electrical resistivity of fly ash significantly influences its collection by ESPs. In general, fly ash with high resistivity is more difficult to collect (and hence, for similar ESPs exhibits lower collection efficiencies) than that with low resistivity. Flue gas temperature has significant influence on resistivity; higher temperatures tend to decrease resistivity, thereby increasing collection efficiency. This is the major design premise behind hot-side ESPs. Although collection efficiency and costs are reduced as a result of decreases in fly ash resistivity, equipment must be sized considerably larger to handle the increased volume (due to thermal expansion) of gas; the cost of such requirements must be taken into consideration. As noted in Section 6.1, fly ash resistivity is dependent upon coal sulfur content; in a similar manner, costs can be related to this parameter.

Flue gas composition can also affect collection efficiency and hence costs for electrostatic precipitation. Here too, the presence of certain species tends to influence resistivity. In general, fly ash resistivity is decreased with increasing coal alkali content, thereby increasing collection efficiency and costs.

Given these variables, a review of the literature indicates that capital costs (in late-1982 dollars) for new electrostatic precipitator installations (both cold and hot-side type) would range from \$30 to 36/kW (64-66). Annual costs for these same systems would range from \$9 to 12/kW per year. Again, wide variability exists among the reported costs in the literature. This is primarily because the capital and operating costs found in the literature, as presented here, attempt to represent a range of plant sizes (200 to 2000 MW), coal sulfur contents, and flue gas compositions.

The capital cost range noted previously for electrostatic precipitator systems consists of direct and indirect capital costs. Direct capital costs include purchased equipment and installation (materials and labor) costs for system battery limits including the basic collector, fans and associated

ductwork. Indirect costs cover such items as field supervision, construction and field expense and engineering and contingency. The annual costs similarly include both direct and indirect costs. Direct operating costs are for items such as operating labor, supervision, maintenance (materials and labor) and required utilities (e.g., power). Indirect costs include such elements as overhead expenses (i.e., payroll and plant overhead), general and administrative costs, taxes, insurance, and capital charges.

The generic costs presented in Figures 6.25 through 6.78 generally agree with the cost estimates developed for the six study sites. Notable differences occur when the design premises for a specific site differ from the design premises that pertain to the generic estimates. Tables 6.7 and 6.8 provide the modular site-specific cost estimates (capital and first year annual costs, respectively) for each of the six study sites, as well as the generic cost estimates that apply to each site. The analysis of these cost data follows:

- Plant Allen - The generic cost estimates for Plant Allen were on the same order as the site-specific estimates, with the exception of costs for the waste transportation module. This is because basalt-lined cast iron and polybutylene slurry piping are used at Plant Allen, and these materials are more expensive than more common piping materials.
- Elrama Plant - The site-specific cost estimate for the processing and handling module was higher than the generic estimates; this is because the Elrama Plant utilizes interim pond dewatering for bottom ash, and the generic costs do not include this option. Since interim ponding is more expensive than other bottom ash processing options, the difference among the generic and site-specific costs is expected. The costs of the waste transport system at Elrama follow a similar trend; this is because the site-specific cost estimate is based on a transport distance of 19 km (12 miles), while the generic cost is based on a shorter distance 1.6 km (1.0 mile).
- Dave Johnston Plant - The Dave Johnston site-specific cost estimates do not agree well with the generic cost estimates. This is because this plant incorporates some expensive options. The fly ash handling system at the Dave Johnston Plant utilizes atypical, expensive Nuva Feeders to transfer ash from the electrostatic precipitator hoppers to the conveying lines. The fly ash is transported via off-the-road trucks that are more expensive than the more typical on-the-road trucks. Additionally, significant landfill excavation is undertaken at this plant. Most plants limited excavation to the quantity of fill needed for reclamation, while additional excavation was required at this plant.
- Sherburne County and Smith Plants - In general, the site-specific and generic costs agreed well for both these plants. Since in both cases, the distance between the plant and disposal area (0.4 km, 0.25 mile) is shorter than that used in the generic cost assessment (1.6 km, 1.0 mile), the waste transportation module costs differ.



TABLE 6.7

## COMPARISON OF SITE SPECIFIC AND GENERIC CAPITAL COST ESTIMATES

|                  |                             | CAPITAL COST (thousand \$)         |                               |               |                           |                             | Waste Placement and Disposal          | Comments <sup>a</sup> |
|------------------|-----------------------------|------------------------------------|-------------------------------|---------------|---------------------------|-----------------------------|---------------------------------------|-----------------------|
|                  |                             | Raw Materials Handling and Storage | Waste Processing and Handling | Waste Storage | Waste Transport           |                             |                                       |                       |
| Allen            | Site Specific Cost Estimate | NA                                 | 5,204                         | NA            | 10,850                    | 25,149                      |                                       |                       |
|                  | Generic Cost Estimate       | NA                                 | 3,300-6,000                   | NA            | 4,800-8,900 <sup>b</sup>  | 17,500-32,400               | Adjustment - coal ash content         |                       |
| Elrama           | Site Specific Cost Estimate | 697                                | 18,820                        | 2,982         | 4,813                     | 7,122                       |                                       |                       |
|                  | Generic Cost Estimate       | 500-900                            | 10,300-19,100 <sup>c</sup>    | 2,200-4,100   | 500-900 <sup>b</sup>      | 4,200-7,400                 | Adjustment - coal ash content         |                       |
| Dave Johnston    | Site Specific Cost Estimate | NA                                 | 6,912                         | 2,512         | 689                       | 4,596                       |                                       |                       |
|                  | Generic Cost Estimate       | NA                                 | 770-1400 <sup>b</sup>         | 1,800-3,400   | 140-260 <sup>b</sup>      | 1,500-2,800 <sup>b</sup>    |                                       |                       |
| Sherburne County | Site Specific Cost Estimate | NA                                 | 21,003                        | NA            | 2,603                     | 38,406 <sup>d</sup>         |                                       |                       |
|                  | Generic Cost Estimate       | NA                                 | 12,100-22,500                 | NA            | 5,600-10,400 <sup>b</sup> | 25,329-49,615 <sup>d</sup>  |                                       |                       |
| Powerton         | Site Specific Cost Estimate | NA                                 | 9,856                         | 6,027         | 1,004                     | 80,075 <sup>e</sup>         |                                       |                       |
|                  | Generic Cost Estimate       | NA                                 | 6,600-11,900                  | 3,300-6,200   | 664-1236                  | 65,700-122,000 <sup>e</sup> | Adjustment - Fly Ash/Bottom Ash Ratio |                       |
| Smith            | Site Specific Cost Estimate | NA                                 | 2,849                         | NA            | 2,442                     | 10,528                      |                                       |                       |
|                  | Generic Cost Estimate       | NA                                 | 1,700-3,200                   | NA            | 1,800-3,300               | 6,300-11,700                | Adjustment - Fly Ash/Bottom Ash Ratio |                       |

<sup>a</sup>Only adjustments to generic costs are noted. Adjustments were made as noted on page 6-54.

<sup>b</sup>See text for discussion of differences among site specific and generic cost estimates.

<sup>c</sup>Excludes cost of interim bottom ash dewatering pond.

<sup>d</sup>Includes the cost of a 0.45 m (1.5 ft) thick clay liner.

<sup>e</sup>Includes the cost of a 3 m (10 ft) thick P02-O-PA00 liner.

Source: Arthur D. Little, Inc. estimates.

TABLE 6.8

## COMPARISON OF SITE SPECIFIC AND GENERIC FIRST YEAR ANNUAL COST ESTIMATES

|                     |                             | CAPITAL COST (thousand \$)            |                                  |                  |                          |                                 | Comments <sup>a</sup>                           |
|---------------------|-----------------------------|---------------------------------------|----------------------------------|------------------|--------------------------|---------------------------------|-------------------------------------------------|
|                     |                             | Raw Materials<br>Handling and Storage | Waste Processing<br>and Handling | Waste<br>Storage | Waste<br>Transport       | Waste Placement<br>and Disposal |                                                 |
| Allen               | Site Specific Cost Estimate | NA                                    | 2,103.7                          | NA               | 2,607.1                  | 4,236.7                         |                                                 |
|                     | Generic Cost Estimate       | NA                                    | 1,100-2,100                      | NA               | 1,400-2,700              | 2,600-5,200                     | Adjustment -<br>coal ash<br>content             |
| Elrama              | Site Specific Cost Estimate | 1,647.2                               | 6,326.3                          | 689.7            | 2,723.6                  | 1,666.6                         |                                                 |
|                     | Generic Cost Estimate       | 1,100-2,100                           | 2,500-5,000 <sup>b</sup>         | 450-1,100        | 600-1,400 <sup>b</sup>   | 1,500-3,000                     | Adjustment -<br>coal ash<br>content             |
| Dave<br>Johnston    | Site Specific Cost Estimate | NA                                    | 1,863.0                          | 579.2            | 374.1                    | 1,000.1                         |                                                 |
|                     | Generic Cost Estimate       | NA                                    | 300-700 <sup>b</sup>             | 400-900          | 200-500                  | 700-1,400                       |                                                 |
| Sherburne<br>County | Site Specific Cost Estimate | NA                                    | 5,237.2                          | NA               | 643.6                    | 6,929.7 <sup>c</sup>            |                                                 |
|                     | Generic Cost Estimate       | NA                                    | 2,905-5,769                      | NA               | 1,500-2,900 <sup>b</sup> | 3,600-8,000 <sup>c</sup>        |                                                 |
| Powerton            | Site Specific Cost Estimate | NA                                    | 2,750.4                          | 1,328.9          | 1,257.3                  | 15,777.9 <sup>d</sup>           |                                                 |
|                     | Generic Cost Estimate       | NA                                    | 2,400-5,100                      | 700-1,500        | 800-1,600                | 10,100-20,600 <sup>d</sup>      | Adjustment -<br>Fly Ash/<br>Bottom Ash<br>Ratio |
| Smith               | Site Specific Cost Estimate | NA                                    | 1,028.2                          | NA               | 420.8                    | 1,594.7                         |                                                 |
|                     | Generic Cost Estimate       | NA                                    | 500-1,000                        | NA               | 500-1,200 <sup>b</sup>   | 1,200-2,300                     | Adjustment -<br>Fly Ash/<br>Bottom Ash<br>Ratio |

<sup>a</sup> Only adjustments to generic costs are noted. Adjustments were made as noted on page 6-54.

<sup>b</sup> See text for discussion of differences among site specific and generic cost estimates.

<sup>c</sup> Includes capital charges for a 0.45 m (1.5 ft) thick clay liner.

<sup>d</sup> Includes capital charges for a 3 m (10 ft) thick POZ-O-PAOS liner.

Source: Arthur D. Little, Inc. estimates.

- Powerton Plant - The generic and site-specific cost estimates for the Powerton Plant agreed with each other.

The site-specific and generic cost estimates are thus fairly similar.

#### 6.2.4 Capital and Operating Cost Estimation Methodology

As previously noted, it is not always necessary to prepare detailed cost estimates where conceptual estimates will suffice. This section explains a simplified method for quickly estimating capital and first year annual costs of both total utility solid waste handling and disposal systems as well as the individual process modules defined here. This method is intended to provide a conceptual cost estimate (with accuracy of plus 30 percent minus 10 percent) for preliminary planning purposes only, and should not be substituted for more detailed estimates prepared from detailed engineering or piping and instrumentation diagrams and site-specific information (e.g., type of terrain, soil conditions, etc.).

Methodology -- The method of estimating waste handling and disposal system costs for a set of specific waste types involves first determining both the capital and annual cost for each and every process module employed in the handling/processing and ultimate disposal of the various waste types (i.e., fly ash, bottom ash and/or FGD waste) generated at the proposed utility plant. This is accomplished with the aid of the appropriate capital and annual cost curves (Figures 6.25 through 6.78). After the costs of each and every individual process module have been estimated for each and every waste type of concern, one then simply adds all of the module costs to arrive at a total waste handling and disposal system cost for the particular plant under consideration. The following example illustrates the use of the cost curves.

Landfill Example: The hypothetical task consists of determining the total capital cost, first year annual cost, and cost per ton of dry waste (including fly ash, bottom ash and FGD waste) for a new 500 MW plant employing fixation of the FGD waste and landfilling of the fixated waste and dewatered bottom ash.

Step 1: Estimate or determine the basic engineering design premises.

- Plant size - 500 MW
- Plant life - 30 years
- Plant load factor - 70 percent
- Heat rate - 10.8 M joules/kWh (10,250 Btu/kWh)
- % S in coal - 2.0 percent
- % Ash in coal - 18.0 percent

- Coal heating value - 24.4 M joules/kg (10,500 Btu/lb)
- Waste quantities generated (dry)
  - fly ash: 195,000 metric tons/year (215,000 tons/yr), collected dry
  - bottom ash: 50,000 metric tons/year (55,000 tons/yr), dewatered in bins, with supernatant recycle
  - FGD waste: 122,000 metric tons/year (135,000 tons/yr)
- Distance to disposal site - 1.6 km (1 mile)
- Land availability - 100 percent
- Liner type - no liner

Step 2: Compare the above given or calculated engineering design premises for the newly proposed plant with those on which the cost curves were based (summarized in Table 6.5). Note any differences that exist between the engineering design premises for the proposed plant and those for which the cost curves are deemed applicable.

|               | <u>Proposed Plant</u> | <u>Cost Curves</u> |
|---------------|-----------------------|--------------------|
| % S in coal   | 2.0                   | 0.5 - 3.5          |
| % Ash in coal | 18.0                  | 12.0 - 15.0        |

Proceed ahead to Step 3, with the understanding that the cost estimates determined from the curves will be somewhat less accurate for this proposed plant because it has a somewhat higher ash content than that on which the cost curves are based. (The sulfur content falls within the range for which capital and annual costs for process modules associated with FGD waste were developed.) One way to reconcile the difference in engineering design premises is to determine a correction factor for this difference in coal ash content and apply this factor to the costs determined from the cost curves. Unfortunately, at this time data are insufficient to determine such correction factors for each individual process module for which cost curves exist. However, correction factors which can be applied to the overall cost of a waste disposal system are available in the open literature (17). These correction factors exist for the following parameters: percent sulfur in coal; percent ash in coal; distance to disposal site; percent of optimum land available; and synthetic liner cost. These factors can be determined from graphs available in Reference 17.

Step 3: Determine the appropriate correction factor for the difference between the coal ash content pertaining to the desired system and the range specified for the cost curves. This factor can be applied to the capital cost estimate for the entire waste handling and disposal system only rather

than for the individual process modules. The correction factor is 1.09. No correction factor is required for the annual cost since these system costs were estimated on the basis of the waste quantity which takes into account the ash content differences.

Step 4: Determine base capital and operating costs for all of the appropriate waste type/process module combinations (summarized in Table 6.9) from the appropriate cost curves.

For all modules, the higher range of the cost curves were taken to determine conservative estimates of both the capital and first-year annual costs (see Table 6.10), since the module options selected for the example in question are complex in nature and typically are represented by the upper regions of the cost curve bands.

Step 5: Calculate corrected capital and annual costs for the overall waste handling disposal system.

$$\begin{aligned}\text{Total Corrected Capital Cost} &= \text{Base Cost} \times \text{Correction Factor(s)} \\ &= (\$31,550,000)(1.09) \\ &= \$34,400,000\end{aligned}$$

Total Corrected First-Year

$$\begin{aligned}\text{Annual Cost} &= \text{Base Cost} \times \text{Correction Factor(s)} \\ &= (\$10,600,000)(1.0) \\ &= \$10,600,000/\text{yr}\end{aligned}$$

$$\begin{aligned}\text{Unit Cost} &= \text{Total Correct First Year Annual Cost} \\ &\quad \div \text{Total Annual Waste Generation} \\ &= \$10,600,000/\text{yr} \div 367,000 \text{ dry metric} \\ &\quad \text{tons/yr} \\ &= \$29/\text{dry metric ton}\end{aligned}$$

### 6.3 Data Gaps

Data gaps related to the economics of coal-fired utility solid wastes (i.e., coal ash and FGD wastes) disposal include both cost information per se, as well as waste properties and disposal requirements that directly impact disposal costs. The most important of these (which may be addressed by government and/or industry initiatives) include the following:

- A general lack of uniformity exists in basic cost assumptions made in engineering cost studies to date. This makes attempts to compare and use such disposal cost estimates from different sources a difficult task, at best.
- A general lack of reliable cost information (capital and operating costs) exists from commercial operation of most types of FGC waste handling and disposal practices.

TABLE 6.9

SUMMARY OF REQUIRED MODULES FOR WHICH COSTS ARE  
ESTIMATED FOR THE LANDFILL EXAMPLE

| <u>WASTE TYPE/PROCESS MODULE</u>      | <u>FIGURE NO.</u>   |                    |
|---------------------------------------|---------------------|--------------------|
|                                       | <u>Capital Cost</u> | <u>Annual Cost</u> |
| ● Dry Fly Ash Handling and Processing | 6.31                | 6.33               |
| ● Fly Ash Storage                     | 6.42                | 6.44               |
| ● Fly Ash Transport                   |                     |                    |
| - dry trucking                        | 6.55                | 6.57               |
| ● Fly Ash Placement and Disposal      |                     |                    |
| - landfill                            | 6.71                | 6.73               |
| ● Bottom Ash Handling and Processing  |                     |                    |
| - wet with recycle                    | 6.37                | 6.39               |
| ● Bottom Ash Transport                |                     |                    |
| - dry trucking                        | 6.58                | 6.60               |
| ● Bottom Ash Placement and Disposal   |                     |                    |
| - landfill                            | 6.74                | 6.76               |
| ● Raw Materials Handling and Storage  | 6.45                | 6.46               |
| ● FGD Waste Handling and Processing   | 6.40                | 6.41               |
| ● FGD Waste Transport                 |                     |                    |
| - dry trucking                        | 6.61                | 6.62               |
| ● FGD Waste Placement and Disposal    |                     |                    |
| - landfill                            | 6.77                | 6.78               |

Source: Arthur D. Little, Inc.



TABLE 6.10  
SUMMARY OF CAPITAL AND OPERATING COSTS FOR FIXATED FGD WASTE/LANDFILL EXAMPLE

| WASTE TYPE/PROCESS MODULE                                         | Capital Cost<br>(\$1000) | Operating Cost<br>(\$1000/year) |
|-------------------------------------------------------------------|--------------------------|---------------------------------|
| • Fly Ash Handling/Processing (dry)                               | \$ 1,700                 | \$ 790                          |
| • Fly Ash Storage (dry, silo)                                     | 3,900                    | 1,050                           |
| • Raw Materials Handling/Storage (fly ash & lime)                 | 840                      | 2,000                           |
| • FGD Waste Handling/Processing (thickening, filtration & mixing) | 14,200                   | 3,120                           |
| • Fly Ash Transport (dry, trucking)                               | 300                      | 530                             |
| • FGD Waste Transport (dry, trucking)                             | 300                      | 520                             |
| • Fly Ash Placement and Disposal (landfill)                       | 3 100                    | 1,650                           |
| • FGD Waste Placement and Disposal (landfill)                     | 3,100                    | 770                             |
| • Bottom Ash Handling/Processing (dewater, with recycle)          | 1,800                    | 750                             |
| • Bottom Ash Transport (dry, trucking)                            | 150                      | 230                             |
| • Bottom Ash Placement and Disposal (landfill)                    | 1,000                    | 350                             |
| TOTAL:                                                            | \$30,390                 | \$11,760                        |

Source: Arthur D. Little, Inc., estimates.

- Insufficient modular division (by process area) of waste disposal system capital and operating costs currently available in the open literature.
- Available capital and operating cost data on mine disposal (open pit and vee-notch) of FGC wastes is lacking, especially for plants with nameplate generating capacities greater than 500 MW.
- No definitive engineering or cost data are available (from conceptual studies or from actual operating systems) on the handling and disposal of FGC wastes from dry sorbent flue gas desulfurization systems.
- Existing physical and engineering properties data on some waste types are not adequate as a basis for developing design requirements needed to prepare reliable estimates of cost-effective disposal systems. Examples include: the disposal of untreated gypsum in dry impoundments; the optimum amount of ash and lime required for fixation of some sulfite-rich FGD wastes; and the potential use of some fixated wastes as liners for dry impoundments of FGC wastes.
- Engineering and cost data (from conceptual studies or from actual operating systems) for the waste handling and disposal of FGD wastes (treated and untreated) in ponds and landfills for all power plant sizes, especially those greater than 500 MW, are insufficient.
- Cost data are lacking concerning the use of interim ponds as a unit processing operation for dewatering coal ash or FGD waste prior to ultimate landfill disposal.
- Actual cost data are lacking regarding the installation of liners (clay, synthetic, etc.) for disposal ponds at various plant sizes.

To summarize, the most important data gap, which is a common thread throughout all of the data gaps, is the lack of detailed engineering/cost breakdown among the existing data (which are limited to begin with) for the disposal of FGC wastes. Consequently, it is very difficult to compare estimated costs with real costs (if they are documented) for a given disposal system or to compare either estimated or real costs for one system with those for another. With only limited cost breakdowns, it is difficult to make comparisons because of uncertainty about what is or is not included within each cost category that is reported. Without such comparisons, it is difficult to recognize which factors are most significant in introducing variations in costs. The result is that certain cost curves, presented here, were based on a limited quantity of data points with less than desired quality. This cost curves presented in Section 6.2 cover a band and, additionally, have an accuracy level of plus 30 percent, minus 10 percent. Another limitation of the cost curves (due to the lack of modularized cost data) is their limited applicability to plants and disposal sites with the same basic range of engineering design premises as those which the curves were based. In other words, for systems with variations from the base case

engineering design premises, one would require correction factors which could be applied to the cost curves presented herein. Unfortunately, however, at this time there are insufficient data to determine such corrections for each individual process module for which cost curves exist. Corrections would be necessary for such case variations as: percentage ash or sulfur in coal, coal heating value, boiler type, FGD system type, transportation distance, and liner costs.

## SECTION 7.0

### ENVIRONMENTAL ASSESSMENT OF COAL ASH AND FGD WASTE DISPOSAL

#### 7.1 MATRIX OF DISPOSAL PRACTICES

The environmental impact of any solid waste disposal practice is determined by the interaction of three factors: waste type, disposal method, and environmental setting. In assessing present and future practices of coal ash and FGD waste disposal, one may identify four basic waste types, three principal disposal methods, and five types of environmental settings of major importance.

##### 7.1.1 Waste Types

As explained in Section 3.3, the four major types of coal ash and FGD wastes are: fly ash or fly ash admixed with other materials, non-fly ash materials, chemically treated FGD wastes, and dry FGD wastes. The chemical and physical characteristics of each waste type were discussed in Section 3.4.

##### 7.1.2 Disposal Methods

Currently, all FGC waste disposal options involve some form of land disposal: pond disposal, interim ponding followed by landfill disposal, and landfill disposal. (Utilization of these waste products is another waste management option. FGC waste utilization is not addressed in this section because it is not as widely practiced as the other options.)

Section 6.1.7 discusses these disposal methods in more detail. They are considered separately for environmental assessment purposes because each involves basically different water management practices, with corresponding implications for the amounts and rates of leachate generation and movement at disposal sites.

##### 7.1.3 Environmental Settings

Three basic environmental settings for solid waste disposal were initially identified for this assessment, based on major differences in climate and geohydrology. These were:

- coastal areas, where surface water and groundwater are directly influenced by the ebb and flow of tides;

- arid areas, characteristic of most of the western U.S., where net evaporation significantly exceeds precipitation; and
- interior areas, characteristic of non-coastal portions of the eastern U.S., where precipitation and evaporation tend to be more balanced than in the west, and where permanent surface water bodies are so abundant that they are close to many waste disposal sites.

Evaluations during this project suggested that two additional types of settings would be useful because of their fundamentally different background water quality characteristics. These are:

- arid areas, highly mineralized, and
- interior areas subject to acid mine drainage.

Both these areas and the coastal setting area tend to have water quality characteristics that, in effect, mask the effects of coal ash and FGD waste leachates. This is because background waters tend to already have high concentrations of certain species that are usually considered contaminants in leachates. More information on all settings and their implications for environmental effects is presented in Section 7.2.

#### 7.1.4 Matrix Combinations

Table 7.1 is a matrix of waste types, disposal methods, and environmental settings. For each setting, the table indicates which combinations of waste type and disposal method apply. For example, pond disposal methods are not applicable to dry FGD wastes, eliminating ten combinations from further consideration. Also, pond disposal of chemically treated FGD wastes is generally not suitable in arid western settings where water availability is a constraint, and in coastal settings where natural valleys that can be converted to impoundments by the disposal practice are lacking.

Table 7.1 also shows disposal/waste/setting combinations for which field-scale and other information is available. Sources other than this project included the Utilities Solid Waste Activity Group (USWAG), EPRI, and DOE. As indicated, some information is available for most of the combinations that are practiced today or likely in the future. For example, full-scale field studies have been conducted in each category of environmental settings except an arid western location that is not highly mineralized. Full-scale field studies have been also been carried out for each major type of disposal practice. The information from these field studies and the extensive data from laboratory and other studies collectively provide a reasonable technical basis for projecting the environmental implications of the other matrix combinations. The environmental implications of the various combinations and their importance in decision-making are discussed in Section 7.3 and in Appendix H.

TABLE 7.1

SUMMARY OF INFORMATION AVAILABLE FOR COMBINATIONS OF WASTE TYPES, DISPOSAL METHODS, AND ENVIRONMENTAL SETTINGS

|                                                                     | Ponding              |             |               | Interim Ponding/Landfilling    |             |               | Landfilling                                                    |             |                         |
|---------------------------------------------------------------------|----------------------|-------------|---------------|--------------------------------|-------------|---------------|----------------------------------------------------------------|-------------|-------------------------|
|                                                                     | Fly Ash <sup>a</sup> | Non-Fly Ash | Processed FGD | Fly Ash <sup>a</sup>           | Non-Fly Ash | Processed FGD | Fly Ash <sup>b</sup>                                           | Non-Fly Ash | Processed FGD           |
|                                                                     |                      |             |               |                                |             |               |                                                                |             |                         |
| COASTAL SETTING                                                     | X <sup>c</sup>       | P           | NA            | X/PA<br>Chisman Cr.<br>(USWAG) | P           | P             | P                                                              | P           | P                       |
| Smith <sup>c</sup>                                                  |                      |             | NA            |                                |             | NA            |                                                                |             | P <sup>e</sup>          |
| ARID WESTERN SETTING -<br>Not Highly Mineralized                    | P                    | P           | NA            | P                              | P           | NA            | X                                                              | P           | P <sup>e</sup>          |
| ARID WESTERN SETTING -<br>Highly Mineralized                        | P                    | P           | NA            | P                              | P           | NA            | X                                                              | P           | P <sup>e</sup>          |
|                                                                     |                      |             |               |                                |             |               | Dave Johnston Milton Young (DOE/EPA)                           |             |                         |
| INTERIOR SETTING -<br>Not Highly Acidic                             | X                    | P           | X             | X/Pd<br>Bailey (USWAG)         | P           | P             | X                                                              | P           | X                       |
| Allen, Sherburne County, Michigan City (USWAG), Wallingford (USWAG) |                      |             | Mansfield     |                                |             | NA            | Powerton, Zuelliger (USWAG), Hunts Brook (USWAG) Dunkirk (DOE) | P           | Conesville (EPRI/USWAG) |
| INTERIOR SETTING -<br>Highly Acidic<br>(mine drainage)              | P                    | P           | P             | P                              | P           | NA            | P                                                              | P           | X <sup>d</sup>          |
|                                                                     |                      |             |               |                                |             |               |                                                                |             | Elrama                  |

Notes: a. Includes co-disposal of fly ash with other wastes.  
b. Includes FGD wastes without fly ash and bottom ash.  
c. Plants for which data and information are obtained are listed in their appropriate positions.  
d. Either the interim pond of landfill aspect of operation studies at field scale, but not both.  
e. Laboratory data only.

Key: X = Data available from full-scale field studies.  
P = Data available from laboratory and/or limited-scale field studies for projection purposes.  
NA = Matrix combination not applicable due to lack of present and future practices.



## 7.2 ENVIRONMENTAL SETTINGS

### 7.2.1 Overview

In any solid waste disposal situation, features of the environmental setting can influence the impact of the disposal practice. Such features are climate, geohydrology, surface water hydrology, and background water quality. Different combinations of these typify each environmental setting considered in this assessment. The environmental settings and their major features are:

- Coastal, typified by influence of tides on groundwater and surface water movement and quality, generally heavy precipitation, and local prevalence of highly pervious soils. This setting prevails along the U.S. Atlantic and Gulf Coasts.
- Arid (not highly mineralized), characterized by little rainfall, high net evaporation, relative absence of aquifers, and various soil types. This setting occurs at isolated locations in the western U.S., much of which is highly mineralized.
- Arid (highly mineralized), with the same climatic and hydrologic features as the previous setting. Other features include alkaline soils, and background water that is relatively high in dissolved chemical species. This setting prevails at many locations in the West.
- Interior (typical), distinguished by: modest rainfall with little difference in net precipitation/evaporation; numerous permanent surface water bodies of varying sizes at most locations, high-quality drinking water supplies in near-surface aquifers; and various soil types. This setting occurs over much of the eastern half of the U.S.
- Interior (acid mine drainage), typified by: same climatic and hydrologic features as the previous setting; water quality and water uses limited by high concentrations of chemical species in mine drainage. This setting is found in the central Appalachian Mountain Region, and at other more scattered locations in the eastern half of the U.S.

The rest of this section discusses the features of these various settings in more detail, with emphasis on the environmental effects of coal ash and FGD waste disposal.

### 7.2.2 Climate and Geohydrology

#### 7.2.2.1 Coastal Environments--

Coastal environments are characterized by low-lying topography, marshes and coastal streams, and nearby salt water bodies with tidal fluctuation of water levels. Waste disposal facilities in coastal environments often act as local groundwater recharge areas, with groundwater and leachate moving in all directions away from the facility. Groundwater and leachate eventually

discharge to the salt water body or to nearby marshes and streams which drain to the salt water body. Leachate flowing through the fresh water aquifer toward the open salt water body is forced upward along the salt water-fresh water interface to discharge at or near the edge of the salt water body.

Rates of leachate movement may be influenced by tidal fluctuations of the nearby salt water body, which change the base levels for groundwater flow and hydraulic gradients away from the waste disposal facility. However, these fluctuations are short-term, and mean-tide conditions will approximate the long-term average leachate movement from the site.

Coastal environments usually experience year-round precipitation and thus year-round leachate movement. But, especially in the Gulf Coast and southeastern Atlantic Coastal Plain, these settings are characterized by intermittent, irregular storm events where substantial precipitation may occur in short periods of time. During such storms, leachate movement may increase due to greater runoff and infiltration in landfills, rising water levels and increased driving forces in ponds, and possible filling of ponds and overtopping of confinement dikes. The storm events and increased leachate movement are short-lived and usually recur once in several years. The probability of a storm occurring at any coastal location can be determined by analyzing historical climatologic data.

#### 7.2.2.2 Arid Western Environments--

In arid environments, potential evapotranspiration exceeds precipitation. Most water losses from a landfill or pond will be to the atmosphere. This may minimize or eliminate percolation or seepage of leachate to groundwater. Precipitation does happen as intense, intermittent events, during which groundwater recharge and leachate movement may occur. But such events may happen only once every several years.

Depth to groundwater and thickness of unsaturated zones are large for waste disposal facilities in upland areas and moderate to small for facilities located on terrace deposits or in alluvial valleys. Directions of groundwater flow are structurally controlled and are towards discharge areas such as streams, where water is lost by runoff, or towards playa lakes, where water is lost by evaporation. Most of the groundwater movement is within alluvial fan complexes, stream terraces, and alluvial valleys.

#### 7.2.2.3 Interior Environments--

Interior environments are usually characterized by humid climates where precipitation slightly exceeds evapotranspiration. Thus, percolation from landfills and seepage from ponds are likely. Although the environment experiences wet and dry periods, precipitation levels tend to be fairly constant, and leachate movement can occur throughout the year. In interior environments characterized by winter ground freezing, water may not infiltrate into a landfill. This will cause a seasonal decrease in percolation. Pond freezing similarly reduces losses to direct evaporation. Depth to groundwater and thickness of the unsaturated zone in interior environments are related to topographic and geologic conditions. Both deep and shallow water table conditions may be present. The direction of groundwater flow in surficial

deposits may be controlled by topography, with groundwater flowing approximately perpendicular to topographic contours. Groundwater will flow to discharge areas such as rivers and streams.

### 7.2.3 Surface Water Hydrology

Most coal-fired power plants in the U.S. are located close to permanent surface water bodies (see Appendix A). This means that many disposal areas for coal ash and FGD wastes are also situated where they may influence or be influenced by a surface water body. These influences can affect leachate generation, leachate movement, and admixing. Surface water hydrology characteristics and their environmental implications for the major settings are summarized below.

#### 7.2.3.1 Surface Water Hydrology - Coastal Settings--

Coastal settings for utility waste disposal represent only those areas subject to tidal influence. Three major types of surface water bodies exist in direct proximity to disposal sites on the U.S. East and Gulf Coasts: estuarine bays, estuarine rivers, and tidal creeks.

The estuarine bays and rivers are usually large water bodies with regular (diurnal), significant, horizontal flows during all seasons. As such, they represent an excellent ultimate mixing, dilution, and dispersion medium for most dissolved species in coal ash and FGD waste leachates. Tidal creeks are also subject to diurnal water exchanges, but their flow volumes are much less than those of the estuarine bays and rivers. Results of several field studies of coal ash landfill leachate dilution in small freshwater streams suggest that flows in even small tidal creeks could have major mixing and dilution impacts on landfill leachates. (See the Powerton site results for this project in Section 5.2 and also References 13 and 67.)

#### 7.2.3.2 Surface Water Hydrology - Arid Settings--

In arid settings, the physical hydrologic characteristics of surface waters in both highly mineralized and less mineralized water bodies are similar. Permanent, natural, surface water bodies are often not present close to disposal sites. Small, flashy temporary streams with seasonal (snow-melt) and event-related flood flows are common, but are of only isolated importance in leachate admixing. A few western disposal sites are located in the alluvial floodplains of the larger permanent rivers. This circumstance can contribute to local water table elevations that affect leachate generation and movement and provide significant ultimate leachate admixing, dispersion, and dilution.

#### 7.2.3.3 Surface Water Hydrology - Interior Settings--

Similar physical hydrologic conditions occur in both the surface waters of the central Appalachian region (where acid mine drainage is most common) and in the other non-tidal interior settings of the Eastern U.S. In the vicinity of any given interior coal ash or FGD waste disposal site, one or more permanent surface water bodies of varying size are likely. These range from very small streams to large lakes and rivers. Field studies in this and other projects strongly suggest that flows in even small, permanent streams

(0.3 to 30 m<sup>3</sup>/s [10 to 100 ft<sup>3</sup>/s) can have a very significant dilution impact upon admixing with coal ash landfill leachates (see Section 5 and References 13 and 67). Comparable data for small streams are not available for pond leachates, but a major dilution potential is again likely.

#### 7.2.4 Water Quality

Typical water quality conditions vary considerably from one environmental setting to another. In three categories (coastal, arid-highly mineralized, and interior-acid mine drainage), many of the major dissolved species concentrations in background waters can be comparable to those in coal ash and FGD waste leachate. The conditions in each setting are discussed below.

##### 7.2.4.1 Water Quality - Coastal Settings--

In background waters of Atlantic and Gulf Coast estuarine settings, concentrations of most major species and some important minor species (i.e., boron) are elevated to levels comparable to those in coal ash and FGD waste leachates. For example, concentrations in bays and rivers of intermediate salinity where coastal disposal sites are located are generally in the same range as that measured at the estuarine (Smith) site studied in this project:

Cl<sup>-1</sup> - 35000 to 15000 mg/l  
SO<sup>-2</sup> - 700 to 2500 mg/l  
B<sup>4</sup> - 1 to 3 mg/l

However, background water concentrations of various trace metals, including As and Se, are typically quite low compared to the upper end of the range reported for coal ash and FGD waste leachates. For example, As and Se background ranges at the estuarine (Smith) site in this project were:

As - 0.6 to 1.2 µg/l  
Se - less than 0.1 to less than 0.3 µg/l

(See Section 5.2 for more information on results from the Smith site.)

Typical water quality conditions in bays and estuaries are also often reflected in near-surface coastal aquifers subject to saline intrusion. This intrusion can cause violation of secondary drinking water standards (i.e., SO<sup>-2</sup> and Cl<sup>-1</sup> >250 mg/l) and so constrains the use of these aquifers for drinking water. Thus, typical water quality conditions in the coastal setting imply that the incremental water quality impacts of the major dissolved species in coal ash and FGD waste leachates may be of little or no concern. For this reason, an assessment of potential adverse effects may have to focus instead on trace metals, such as As and Se, and on non-drinking water standards (i.e., those established to protect aquatic life).

##### 7.2.4.2 Water Quality - Arid Setting, Not Highly Mineralized--

The arid and generally alkaline nature of much of the western U.S. makes much of the surface and groundwater quality of the area "highly-mineralized," as discussed below. But even in these highly-mineralized settings, surface



water and groundwater quality is relatively free from elevated concentrations of dissolved species. For example, the results of this project for the Johnston site, which is generally highly-mineralized, also showed some quite low concentrations of dissolved species at isolated sampling locations:

$\text{SO}_4^{-2}$  - 38 to 144 mg/l  
 $\text{Cl}^-$  - 2 to 8 mg/l  
B - 0.01 to 0.06 mg/l

Such levels are well below applicable secondary drinking water standards (see Section 7.2.4.1) and other potential constraints on use (i.e., B in water for agricultural use: 0.750 mg/l). Trace metal concentrations in these background waters are also typically below drinking water standards. Ranges at the Johnston site were:

As - less than 0.5  $\mu\text{g/l}$  (standard = 50  $\mu\text{g/l}$ )  
Se - 1.3 to 5.7  $\mu\text{g/l}$  (standard = 10  $\mu\text{g/l}$ )

(See Section 5.2 for more results on the Johnston site.)

The surface and groundwater quality of arid western settings that are not highly mineralized can usually support a full range of uses, including drinking water. Thus, the potential for incremental impacts by both major and trace dissolved species in coal ash and FGD waste leachates should be considered.

#### 7.2.4.3 Water Quality - Arid Setting, Highly Mineralized--

Like the coastal settings discussed above, concentrations of at least some major dissolved species in the background surface and groundwaters of highly-mineralized arid western settings are often higher than secondary drinking water standards. Such background waters have been studied as part of this and other projects (see results on the Johnston site in Section 5.2 and also Reference 68). Measured ranges include:

$\text{SO}_4^{-2}$  - 600 to 1300 mg/l (standard = 250 mg/l)  
B - 0.7 to 5.1 mg/l (criterion = .750 mg/l)

Also like the coastal setting, concentrations of major trace metals tend to be below applicable drinking water standards. Measured ranges of As and Se in background groundwaters from the same two projects (see Reference 68 and Section 5.2) were:

As - 1.0 to 8.1  $\mu\text{g/l}$  (standard = 50  $\mu\text{g/l}$ )  
Se - less than 0.8 to 2.5  $\mu\text{g/l}$  (standard = 10  $\mu\text{g/l}$ )

Because of high levels of major species, water in arid, highly mineralized settings may not be suitable for drinking water. This places primary focus on potential trace metal impacts for other than drinking water uses (i.e., aquatic life, agriculture).

#### 7.2.4.4 Water Quality - Interior Settings (Typical)--

As in the not highly mineralized arid setting, background surface water and groundwater in interior settings free from major contamination generally have both major and trace species concentrations that allow the full range of water uses, including drinking water. For example, concentrations of selected species in background surface and groundwater samples from the typical interior settings sampled in this project (Powerton, Allen and Sherburne County) were:

|                    |   |                                                      |
|--------------------|---|------------------------------------------------------|
| $\text{SO}_4^{-2}$ | - | 0.5 to 84 mg/l (standard = 250 mg/l)                 |
| B                  | - | less than 0.005 to 0.63 mg/l (criterion = .750 mg/l) |

Ranges of selected trace metals (As, Se) in background waters at the same sites were:

|    |   |                                                                       |
|----|---|-----------------------------------------------------------------------|
| As | - | less than 0.2 to 0.6 $\mu\text{g/l}$ (standard = 50 $\mu\text{g/l}$ ) |
| Se | - | less than 0.3 $\mu\text{g/l}$ (standard = 10 $\mu\text{g/l}$ )        |

(See Section 5.2 for further results from the Powerton, Allen and Sherburne County sites.)

Because these waters have low background concentrations and can support a full range of uses, the initial focus for assessing impacts on water quality in interior settings is on both the major and trace dissolved species in coal ash and FGD waste leachates.

#### 7.2.4.5 Water Quality - Interior Setting (Acid Mine Drainage)--

The poor quality of waters affected by acid mine drainage causes more potential use constraints than those experienced by other settings. Water bodies so affected are limited in their ability to support recreational fisheries as well as drinking water needs. For example, the ranges of selected species concentrations measured in groundwater believed to be affected only by acid mine drainage at the representative site (Elrama) were:

|                    |   |                                             |
|--------------------|---|---------------------------------------------|
| $\text{SO}_4^{-2}$ | - | 85 to 1900 mg/l (standard = 250 mg/l)       |
| B                  | - | 0.015 to 0.407 mg/l (criterion = .750 mg/l) |
| pH                 | - | 4.5 to 6.1 (standard = 6.5 - 9.5)           |

Ranges of the two important trace metals in the same groundwaters were:

|    |   |                                                                |
|----|---|----------------------------------------------------------------|
| As | - | 0.4 to 8.6 $\mu\text{g/l}$ (standard = 50 $\mu\text{g/l}$ )    |
| Se | - | less than 0.3 $\mu\text{g/l}$ (standard = 10 $\mu\text{g/l}$ ) |

(See Section 5.2 for more information on the Elrama site.)

As in other settings where background water shows elevated concentrations of species important in coal ash and FGD waste leachates, the implication for assessments here is that drinking water uses may already be precluded and that only the limited number of species affecting other uses should be considered.



## 7.3 MECHANISMS OF ENVIRONMENTAL EFFECTS

### 7.3.1 Overview

The environmental (water quality) factors discussed in Section 7.2 were assessed for each combination of waste type, disposal method and environmental setting, as shown in Table 7.1. Results are documented in Appendix H. The environmental findings at each of the six field sites were presented in Section 5.2. Results from both efforts are summarized below in terms of their implications for environmental (water quality) assessment and decision-making for prospective coal ash or FGD waste disposal practices.

Sections 7.3.3 through 7.3.8 summarize the effects and decision-making implications of six categories of the potential combinations shown in Table 7.1. These categories were selected because each shows fairly unique environmental effects and decision-making implications. The categories are:

1. Coastal setting, various waste types and disposal methods;
2. Arid setting - highly mineralized, various waste types and disposal methods;
3. Arid setting - not highly mineralized, pond disposal;
4. Arid setting - not highly mineralized, landfill disposal;
5. Interior setting - typical, various waste types and disposal methods; and
6. Interior setting - acid mine drainage, various waste types and disposal methods.

In three categories (1, 2 and 6), typical background water quality requires less emphasis on the impact of major dissolved species (i.e., sulfate) on water quality. Initial focus instead should be on the less frequently problematic trace metals in the waste leachate. In Category 2, potentially adverse effects are mitigated by the likely lack of physical proximity between the disposal site and usable receiving waters, and by the general minimization of leachate generation and movement in arid settings. Categories 3 and 4 share some of these advantages, although greater water movement generally accompanies the pond versus landfill disposal method. Finally, in Category 5, few, if any, potential assessment considerations can be ruled out a priori, and the decision process can regularly require in-depth analysis of each major mechanism of environmental (water quality) effects - leachate generation, movement, admixing, and attenuation.

### 7.3.2 Implications of Waste Type

Despite the differences in chemical and physical properties of coal ash and FGD wastes (see Section 3.4), there are also some basic similarities that result in common major effects and decision-making implications. Specifically, all four major waste categories (fly ash, bottom ash/FGD waste,

chemically treated FGD wastes, and dry FGD wastes) share three characteristics:

- Despite variations in physical properties such as permeability, all these wastes are expected to release measurable leachates into the adjacent environment within a few years of placement by any prevalent unlined disposal method. (The quantity of measurable leachate release can vary significantly from one site to another.)
- Leachate from each waste type is expected to exceed Secondary Drinking Water Standards for at least one major dissolved species (sulfate). Fewer wastes have concentrations of multiple species in excess of these standards. For example, excess levels of chloride tend to be restricted to FGD wastes.
- Leachates sampled under actual or simulated field conditions from each waste type are highly variable in trace metal concentrations, but a minority of cases have exhibited concentrations in excess of the Primary Drinking Water Standards.

#### 7.3.2.1 Fly Ash Wastes--

Fly ash wastes, including separately disposed fly ash and fly ash co-disposed with bottom ash or FGD waste, have been studied at field scale in each major environmental setting, as placed by several major disposal methods. The ability of these wastes to release measurable leachate into the adjacent environment has been documented under at least the following circumstances:

- pond disposal of fly ash and bottom ash, coastal setting (Smith site studied during this project; see Section 5.2);
- landfill disposal of fly ash following interim ponding, coastal setting (Chisman Creek site results reported in Reference 13);
- pond disposal of fly ash and bottom ash, interior settings (Allen site studied during this project, as described in Section 5.2; and Wallingford site results reported in Reference 13);
- pond disposal of fly ash and FGD waste, interior setting (Sherburne County site studied during this project; see Section 5.2);
- interim pond disposal of fly ash, interior setting (Bailly site results reported in Reference 13);
- landfill disposal of fly ash and bottom ash, western setting (Johnston site studied during this project; see Section 5.2);
- landfill disposal of fly ash and FGD waste, western setting (Reference 5);

- landfill disposal of fly ash and bottom ash, interior setting (Powerton site studied during this project, as described in Section 5.2; and Zuellinger and Hunts Brook sites described in Reference 13).

Given this wide range of conditions, it is reasonable to anticipate that these wastes will release measurable leachate into adjacent surroundings in the absence of additional physical barriers (i.e., site liners).

In this project, sulfate concentrations in the in-situ samples of fly ash waste interstitial liquids exceeded the secondary drinking water standard of 250 mg/l at all sites where fly ash wastes were disposed of (Smith, Johnston, Powerton, Allen and Sherburne County). Concentrations measured ranged from 320 to 1780 mg/l (Secondary Drinking Water Standard = 250 mg/l). At three sites (Smith, Johnston and Sherburne County), sulfate concentrations greater than 1000 mg/l were measured in the in-situ, interstitial waste liquors. Other major dissolved species in fly ash waste leachates may be expected to exceed Secondary Drinking Water Standards in a minority of cases. Such species includes chloride (especially where co-disposal with FGD wastes is involved--see Sherburne County site results for this project and also References 2 and 3) and manganese (67). The Water Quality Criterion for boron of .750 mg/l, is based on certain crop sensitivities to levels in irrigation water (6), is regularly exceeded by fly ash waste interstitial waters, as demonstrated at all five fly ash waste sites in this project.

Trace metal species that receive primary emphasis in assessments of coal ash and FGD waste leachates include arsenic and selenium, because of their potential to exceed applicable National Interim Primary Drinking Water Standards (50 µg/l for As and 10 µg/l for Se) (2,3). Results of this project support previous indications that fly ash waste leachates contain concentrations of these species that exceed the Standards in a minority of cases. At the four fly ash waste sites where Se analyses of waste interstitial waters were performed (Powerton, Smith, Allen and Sherburne County), the range of measured values was less than 0.1 to 6.6 µg/l, all below the 10 µg/l Standard, although at one site (Sherburne County), pond supernatant contained ~250 µg/l of Se. Arsenic values measured in the same waste interstitial waters were below the 50 µg/l Standard at three of the four sites, ranging from 0.7 to 11 µg/l, but exceeded the Standard at the fourth site (Allen), ranging from 1550 to 2425 µg/l.

#### 7.3.2.2 Non-Fly Ash Wastes (Bottom Ash, FGD Wastes)--

Disposal of bottom ash and/or FGD waste separately from fly ash is fairly uncommon, and data from full-scale studies on leachate generation and chemical quality are lacking. The effects of these wastes must be estimated based on available laboratory and pilot scale field work.

Data indicate that the in-situ permeability of bottom ash alone is generally greater than or equal to that of fly ash (2,3). This means that bottom ash disposal areas will physically release leachate to the adjacent environment if additional physical barriers, such as liners or leachate collection systems, are not present. However, leachates from bottom ash usually do not have trace elements of concern in such concentration as do fly

ash leachates. Unstabilized FGD wastes also release leachate, based on limited data comparing them to stabilized FGD wastes which release leachate in field applications (2,3,5; Conesville site results reported in Reference 1, and Elrama site results in this project).

Like fly ash, bottom ash liquors at field-scale have been shown in at least one case (Sherburne County site) to contain sulfate and boron levels that exceed the Secondary Drinking Water Standards and water quality criteria. The separate bottom ash disposal area characterized at Sherburne County showed sulfate levels of 2200 to 2300 mg/l and boron at about 20 mg/l. Although this data base is small, more extensive laboratory and limited field-scale studies of unstabilized FGD wastes indicate that the same species, in addition to chloride, would likely exceed the secondary drinking water standards and water quality criteria in full-scale applications (2,3,6,15).

Data suggest that As and Se concentrations in ash-free, unstabilized FGD waste and bottom ash liquor may exceed the Interim Primary Drinking Water Standards, but in an even smaller minority of cases, (and perhaps not at all for bottom ash) (2,3,15,16,69,70). Water leaching of Se concentrations higher than 10 µg/l from ash-free FGD wastes has been documented in at least two laboratory studies of wastes produced by different scrubbing techniques (15,16,70).

#### 7.3.2.3 Chemically Treated FGD Wastes--

Results of this project at the Elrama site and from other limited field-scale studies (15,16,69) indicate that chemical treated wastes generally have similar leaching potential as those in the preceding two categories. The chemically treated FGD waste landfill studied at the Elrama site showed leachate release to the adjacent environment, as did other treated FGD wastes in limited field-scale pond disposal studies (2,15). Sulfate concentrations in the in-situ waste interstitial waters at Elrama ranged from 700 to 2020 mg/l, and some chloride values exceeded the 250 mg/l secondary drinking water standard, generally by about a factor of two. Similar results (with higher chloride levels) were reported for the limited field-scale studies (15,16,69). Se levels at Elrama were less than 4 µg/l, but As levels measured in the in-situ waste site interstitial waters ranged from 5.9 to 266 µg/l; the upper end of this range is roughly five times the Interim Primary Drinking Water Standard. As and/or Se values greater than limits set by standards have also been reported for some treated FGD wastes, but the number of wastes tested seems too small to determine whether such concentrations could be expected in most field situations (15,16,68,69,70).

#### 7.3.2.4 Dry FGD Wastes--

Characterization data on dry FGD wastes are available only from laboratory-scale studies. These data suggest that such wastes should have similar characteristics as the other types of wastes discussed above (3,70).

In work sponsored by EPRI, laboratory permeability values for fourteen different dry scrubbing wastes produced at nine plants ranged from  $1.6 \times 10^{-4}$  cm/sec to  $2.7 \times 10^{-7}$  cm/sec, a range comparable to that for the other coal ash and FGD wastes shown to allow leachate release at full field scale (2,18).



Sulfate concentrations in water extracts of dry FGD wastes have been reported to range from 1100 to 4200 mg/l, placing these leachates in excess of the Secondary Drinking Water Standard (2).

One EPRI study reported that the Interim Primary Drinking Water Standard for As was exceeded in neutral extracts of wastes at two of the nine plants studied and for Se at four of the nine (18). Comparable standards for Cr and Ba were also exceeded (70). These very preliminary results again indicate that, in a minority of cases, selected trace metals may be leached in concentrations of potential concern.

### 7.3.3 Implications of Disposal in Coastal Settings - Various Waste Types and Disposal Methods

#### 7.3.3.1 Practices in this Category--

As described in Section 7.2, the coastal setting includes those portions of the U.S. East and Gulf Coasts subject to the regular (diurnal) influences of tidal action on surface waters and near-surface aquifers. (Other non-tidal portions of the traditionally defined "coastal plain" are considered in this study as part of the interior setting.)

Of the possible combinations of waste types, disposal methods and environmental settings shown in Table 7.1, eight waste/disposal method combinations are applicable to the coastal setting and have similar decision-making implications. These combinations are (13):

- Fly ash waste, pond disposal - By far the predominant practice at U.S. East and Gulf Coast plants, and expected to remain important in the future.
- Bottom ash or ash-free FGD waste, pond disposal - Unlikely to prevail in the future.
- Fly ash waste, interim pond-landfill disposal - Not common, and unlikely to become so.
- Fly ash waste, landfill disposal - An uncommon practice that is expected to become more prevalent as additional coastal plants convert to coal for their primary fuel source.
- Bottom ash or ash-free FGD waste, landfill disposal - Not commonly applied and unlikely to become more prevalent.
- Chemically treated FGD waste, landfill disposal - Not now practiced but may be in the future.
- Chemically treated FGD waste, interim pond/landfill disposal - Not applied at full scale in the coastal setting and unlikely to prevail in the future.

- Dry FGD waste, landfill disposal - In the early stages of commercialization; may be applied in the future.

#### 7.3.3.2 Status of Documentation--

Field studies of full-scale coal ash or FGD waste disposal in the coastal setting have been reported for at least three sites. During this project, a combined fly ash/bottom ash disposal pond in the Florida panhandle was evaluated at the Smith site. Results are summarized in Section 5.2.6. More limited studies of final land disposal of interim-ponded fly ash in coastal Virginia were reported recently by USWAG (13). At a site in coastal Maryland, initial investigation by the State led to a planned program of more detailed field studies (71).

#### 7.3.3.3 Summary of Environmental Implications--

The various coal ash and FGD wastes should have the following generic environmental implications when disposed of by any of the major methods in a tidally-influenced coastal setting:

- Leachate quality should have concentrations of major species (i.e., sulfate, boron) comparable to those of tidally-influenced background surface waters and near-surface aquifers. This was apparent and expected in the circumstances encountered at the Smith site in this project, where estuarine water is used as a regular or periodic waste transport medium.
- In some cases, concentrations of certain trace metals (i.e., As, Se) in the leachate may be higher enough than that of the background waters to be of potential concern (see Sections 7.2 and 7.3.2 above). For example, results for the coastal Virginia site showed disposal-related elevated levels of Se and V (13), and vegetation analyses at the coastal site in Maryland prompted concern over As and Se levels (71).
- The prevalence of heavy precipitation and pervious soils in the coastal setting can promote rapid leachate generation and movement. Off-site leachate migration was observed at both of the pond and landfill coastal coal ash disposal sites that have been studied (see Reference 13 and results for Smith site in Section 5.6.5 of this report).
- Opportunities for extensive admixing with tidally-influenced surface waters and near-surface aquifers can eliminate or reduce substantially the possibility of adverse impacts from major dissolved leachate species. As noted in Section 7.2, concentrations of major dissolved species in coastal settings are generally such that surface water and near-surface aquifers are not available as untreated drinking water supplies. Secondary Drinking Water Standards often have little or no applicability, and the relevant criteria may be limited to those designed to protect aquatic life in surface waters. Major dissolved species in coal ash and FGD waste leachates are generally of little concern with respect to the marine aquatic life criteria, and would be



of essentially no concern in a situation where admixing with tidally-influenced surface waters prevailed (6).

- Chemical attenuation of trace metals can be expected in some but not all of the prevalent situations. Among the many soil types found in coastal settings are coarse sands of relatively low organic content and exchange capacity and, by implication, less anticipated ability to attenuate trace metals (see Appendix A). Yet this setting also includes wetland and other areas where organically enriched soils and some silts and clays prevail. These latter soils should be able to attenuate significantly trace metals. While field-scale data are sparse, significant soil attenuation of at least one trace metal species (V) was observed at a Virginia coastal coal ash disposal site, (see Chisman Creek results reported in Reference 13) and some of the soils at the coastal (Smith) site in this project had significant, though pH-dependent, attenuation capacity (see Section 5.2.6 and Appendix F).

Thus, opportunities for adverse effects of various disposal methods for coal ash or FGD waste in the coastal setting seem restricted to those few cases where elevated leachate concentrations of one or more trace metals (i.e., As or Se) may prevail, and specifically to the even more unlikely situation where a relative absence of both physical admixing and chemical attenuation of the elevated leachate metals concentrations could cause concentrations in an ambient surface water body to reach threshold levels harmful to resident aquatic life. If the latter circumstance were expected, mitigation could be achieved by selecting an alternative disposal site, or by placing an appropriately impermeable and/or chemically attenuative soil as a site liner.

#### 7.3.4 Implications of Disposal in Arid, Highly-Mineralized Settings - Various Waste Types and Disposal Methods

##### 7.3.4.1 Practices in this Category--

As described in Section 7.2, the arid, highly-mineralized setting is typical of many locations in the western U.S. This setting is distinguished from other arid settings by high concentrations of dissolved minerals in surface waters and near-surface aquifers, generally in excess of one or more secondary drinking water standards.

Of the possible combinations of waste types, disposal methods and environmental settings shown in Table 7.1, nine waste/disposal method combinations are applicable to the arid, highly-mineralized setting and have similar decision-making implications. These combinations (1) are:

- Fly ash waste, pond disposal - A fairly common practice that is expected to become less prevalent as users turn to dry disposal.
- Fly ash waste, interim pond-landfill disposal - An uncommon method that is likely to remain so, especially as dry disposal becomes more widely used.

- Fly ash waste, landfill disposal - An important practice that should become even more popular, since it is the major applicable combination in the West.
- Bottom ash or ash-free FGD waste, pond disposal - A combination that is practiced but not expected to prevail in the future.
- Bottom ash or ash-free FGD waste, interim pond-landfill disposal - A fairly uncommon practice that is unlikely in the future.
- Bottom ash or ash-free FGD waste, landfill disposal - A relatively uncommon practice, expected to remain so because of the likelihood of co-disposal with fly ash.
- Dry FGD waste, landfill disposal - In the early stages of commercialization in the West and likely to become more prevalent.
- Chemically treated FGD waste, landfill disposal - Does not occur now but may be practiced at a few locations in future.
- Chemically treated FGD waste, interim pond-landfill disposal - Not practiced presently and unlikely in the future.

#### 7.3.4.2 Status of Documentation--

Field studies of full-scale coal ash or FGD waste disposal in arid, highly-mineralized settings have been reported for at least two sites. As part of this project, a combined fly ash/bottom ash landfill in a Wyoming alluvial floodplain was evaluated in detail, and results are summarized in Section 5.2.3 (Dave Johnston site). A research/demonstration (and later full-scale) operation involving disposal of fly ash and fly ash-scrubbed FGD waste by landfill in a North Dakota surface mine has also been evaluated (5). Measured background water quality conditions in both these areas generally fell into the category of "highly-mineralized," as described in Section 7.2.

#### 7.3.4.3 Summary of Environmental Implications--

The various coal ash and FGD wastes, should have the following generic environmental implications when disposed of by any of the major methods in an arid, highly-mineralized setting:

- Both background waters and leachate are likely to have concentrations of major dissolved species (i.e., sulfate) well above limits set by Secondary Drinking Water Standards. This was apparent at both field study sites. At the Johnston site studied during this project, major species concentrations were extremely similar in waste and background waters. At the other site, the highly alkaline FGD wastes produced some leachates with measurably higher concentrations of certain major species than the high background levels (68,5).
- In some cases, leachate concentrations of certain trace metals (i.e., As, Se) may be higher enough than those of the background waters to approach or exceed applicable standards and thresholds. For example,

at the North Dakota site, alkaline fly ash-related leachate concentrations of As and Se in groundwater were significantly higher than the concentrations in the highly mineralized background waters, and above limits set by the Interim Primary Drinking Water Standards (68,5).

- Minimal precipitation, high net evaporation, and the general absence of significant near-surface groundwater generally reduce opportunities for leachate generation and movement. In many arid settings, receiving waters are so removed that waste leaching is confined to the unsaturated zone near the waste deposit. Exceptions do occur, as at the western (Johnston) site studied in this project. In this case, the waste was deposited by excavation in an alluvial floodplain, and this caused contact between the waste and locally high groundwater. But even at this unusual type of western site, the generally arid setting minimizes and restricts leachate generation and movement to the immediate vicinity of the waste deposit, especially when dry disposal methods are practiced.
- Even if disposal conditions allow leachate to contact receiving waters, its admixing with the highly-mineralized background water would result in few, if any, incremental effects from major dissolved leachate species, such as sulfate. However, the highly mineralized receiving waters are often of limited importance as untreated drinking water supplies. On the other hand, they may continue to be valued for agricultural purposes and for support of aquatic life. The latter are uses for which the typical reported concentrations of leachate major species would have few, if any, negative implications upon admixing (6).
- Chemical attenuation of trace metals can be expected in some, but not all of the prevalent situations. The highly mineralized arid setting has a broad range of soils with varied physical and chemical attenuative capacities (see Section 7.2). At one field site, waste alkalinity together with a lack of attenuation may have contributed to the elevated As levels measured at some near-downgradient sampling locations (68). Some of the soils at the arid, highly mineralized Johnston site studied in this project showed minimal attenuation capacity for most trace metals of interest.

In summary, the adverse effects of coal ash or FGD waste disposal by various methods in arid high-mineralized settings should be minimal and restricted to those cases where wastes with elevated levels of one or more trace metals (i.e., As) are placed so close to useful receiving waters that elevated receiving water concentrations develop in spite of the aridity of the setting. If such conditions were expected, the problem could be avoided by choosing a better-situated alternative disposal sites (which should be available even within a very small geographic area). In the unlikely event that no alternative site was feasible, the effects could be mitigated by placing an appropriately impermeable and/or chemically attenuative soil as a site liner.

### 7.3.5 Implications of Pond Disposal in Arid, Not Highly-Mineralized Settings - Various Waste Types

#### 7.3.5.1 Practices in this Category--

As described in Section 7.2, the arid, not highly-mineralized setting includes many isolated locations in the western U.S. It is distinguished from the highly-mineralized areas by the prevalence of superior water quality, adequate to support a broader spectrum of uses, including use as a potable water supply without further treatment.

Of the combinations of waste types, disposal methods and environmental settings shown in Table 7.1, five ponding and interim pond-landfill disposal methods can be practiced in the arid, not highly-mineralized setting and have similar decision-making implications. These combinations (13) are:

- Fly ash waste, pond disposal - Common practice at present but expected to become less prevalent as dry disposal gains acceptance.
- Fly ash waste, interim pond-landfill disposal - Uncommon and expected to remain so, especially as dry disposal becomes more prevalent.
- Bottom ash or ash-free FGD waste, pond disposal - An existing practice that is unlikely to become prevalent in the future.
- Bottom ash or ash-free FGD waste, interim pond-landfill disposal - Uncommon, and unlikely for the future.
- Chemically treated FGD waste, interim pond-landfill disposal - A combination that, while possible, does not occur and is unlikely in the future.

#### 7.3.5.2 Status of Documentation--

Applicable data from full-scale field studies are not available. Useful data can be obtained from other studies of full-scale practices in arid settings (5,68), as well as results from the Johnston site in this project. This information, along with the extensive literature on laboratory and limited field-scale investigations of coal ash and FGD waste disposal (2,3), provides a basis for projecting the likely environmental implications for this setting.

#### 7.3.5.3 Summary of Environmental Implications--

The various coal ash and FGD wastes should have the following generic environmental implications when disposed of by ponding (including interim ponding) in an arid, not highly-mineralized setting:

- Leachate concentrations of certain major chemical species (i.e., sulfate) can be higher enough than those of background waters to be of potential concern with respect to Secondary Drinking Water Standards. (See Sections 7.2 and 7.3.2.)

- In a few cases, leachate may have sufficiently higher concentrations of certain trace metals (i.e., As, Se) than those of the background waters to be problematic (see Sections 7.2 and 7.3.2).
- Pond head can promote leachate generation and movement. However, the combination of minimal precipitation, high net evaporation, and the general absence of significant near-surface groundwater generally reduce the extent of leachate plume movement. Based on results of full-scale pond studies in the less arid areas of the eastern U.S. (see summaries for the Smith, Allen, and Sherburne County sites in Section 5), measurable leachate will likely remain within the immediate vicinity (i.e., a few hundred meters maximum) of typical western disposal ponds during their operating lifetimes.
- Chemical attenuation of trace metals can be expected in some, but not all, of the prevalent situations (see Section 7.3.4.3).

The adverse effects of coal ash or FGD waste disposal by ponding or interim pond-landfill methods in arid, not highly-mineralized settings would occur only if elevated concentrations of major and (less likely) trace metal species could reach an immediately adjacent drinking water supply or small, usable surface water body. Such conditions are unlikely, but if they were expected, the problem could be avoided by selecting an alternative disposal site or using an impermeable and/or chemically attenuative soil as a site liner.

### 7.3.6 Environmental Implications of Landfill Disposal in Arid, Not Highly Mineralized Settings - Various Waste Types

#### 7.3.6.1 Practices in this Category--

Landfill disposal in an arid, not highly mineralized setting is placed in a separate category than pond disposal because of the differences in water movement associated with each practice. Since a landfill disposal site experiences less water movement than a waste pond, the environmental effects of landfilling are generally more acceptable, as explained below.

Four types of waste are suitable for landfill disposal in the arid, not highly mineralized setting:

- Fly ash waste - A prevalent practice that is expected to become even more popular.
- Bottom ash or ash-free FGD waste - Fairly uncommon and likely to remain so since these wastes are usually co-disposed with fly ash.
- Dry FGD wastes - In the early stages of commercialization in the West and expected to become more prevalent.
- Chemically treated FGD waste - Not now practiced but may be in the future.



#### 7.3.6.2 Status of Documentation--

Data from full-scale studies of landfill disposal in this setting are not available, although useful information can be obtained from other investigations of full-scale practices in arid settings (5,68), as well as results from the Johnston site in this project. This information, along with the extensive literature on smaller-scale studies of coal ash and FGD waste disposal (2,3), provides a basis for projecting the likely environmental implications for this setting.

#### 7.3.6.3 Summary of Environmental Implications--

The environmental implications of disposal practices in this setting are similar to those described in Section 7.3.5.3, with one major difference. In the case of landfill disposal, the combination of dry disposal methods, minimal precipitation, high net evaporation, and the absence of significant, near-surface groundwater makes opportunities for leachate generation and movement even more unlikely.

#### 7.3.7 Environmental Implications of Disposal in Typical Interior Settings - Various Waste Types and Disposal Methods

##### 7.3.7.1 Practices in this Category--

The range of typical interior settings includes most of the eastern half of the U.S., excluding only those areas subject to tidal influence (coastal) and acid mine drainage. Ten of the waste/disposal method combinations listed in Table 7.1 apply to this setting and have essentially similar decision-making implications. These are:

- Fly ash waste, pond disposal - The most dominant practice in the eastern U.S. and expected to remain important, although many new facilities will use dry disposal.
- Fly ash waste, interim pond-landfill disposal - A common practice that will become less prevalent as users turn to simpler practices.
- Fly ash waste, landfill disposal - A frequent practice that should become even more important.
- Bottom ash or ash-free FGD wastes, pond disposal - A fairly popular practice for bottom ash that is also used for FGD wastes. It will remain important for bottom ash, but may decline for FGD wastes in favor of fly ash co-disposal methods.
- Bottom ash or ash-free FGD wastes, interim pond-landfill disposal - A prevalent practice for bottom ash, but rare for FGD wastes; expected to be less common in the future.
- Bottom ash or ash-free FGD wastes, landfill disposal - Uncommon and unlikely to become more important because of the greater likelihood of co-disposal with fly ash.



- Chemically treated FGD waste, pond disposal - Practiced at only two sites and likely to remain fairly rare.
- Chemically treated FGD waste, interim pond-landfill disposal - Not practiced at present; feasible, but unlikely for the future.
- Chemically treated FGD waste, landfill disposal - Recently initiated at several interior sites and likely to become much more prevalent.
- Dry FGD waste, landfill disposal - In the early stages of commercial application and expected to become more important.

#### 7.3.7.2 Status of Documentation--

A substantial data base is available for this setting. Field studies of full-scale coal ash or FGD waste disposal in typical interior settings have been reported for at least eleven sites:

- Combined fly ash and bottom ash ponding in North Carolina, studied at the Allen site as part of this project.
- Combined fly ash and FGD waste ponding, and separate bottom ash ponding, in central Minnesota, studied at the Sherburne County site as part of this project.
- Combined fly ash and bottom ash landfill in west-central Illinois, studied at the Powerton site as part of this project.
- Coal ash ponding in the Indiana Lake Plain and in Connecticut (Michigan City site, Wallingford site results, respectively, reported in Reference 13).
- Ponding of chemically treated FGD waste on the Pennsylvania/West Virginia border (72).
- Ponding of interim pond-landfill disposal of coal ash in the Indiana Lake Plain (Bailly site results reported in Reference 13).
- Landfill disposal of coal ash in Pennsylvania, northern New York and Connecticut (Zuelliger site, Chisman Creek site, and Dunkirk site results, respectively, reported in Reference 13).
- Landfill disposal of chemically treated FGD wastes in Ohio (Conesville site results reported in Reference 13).

#### 7.3.7.3 Summary of Environmental Implications--

The various coal ash and FGD wastes will likely have the following generic environmental implications when disposed of by any of the major methods in typical interior settings:

- Leachate concentrations of certain major chemical species (i.e., sulfate) will be higher enough than background levels to approach or

exceed Secondary Drinking Water Standards (see Sections 7.2 and 7.3.2). The Secondary Drinking Water Standard for sulfate was exceeded in waste interstitial waters at all three typical interior sites in this project (Powerton, Allen, and Sherburne County), and major dissolved species were of concern at other sites (72; Zuellinger site and Hunts Brook site results reported in Reference 13).

- In some cases, leachate concentrations of certain trace metals (i.e., As, Se) in groundwater may be problematic (see Sections 7.2 and 7.3.2). Waste interstitial waters had As or Se concentrations in excess of the Interim Primary Drinking Water Standards at two of the three typical interior study sites in this program. Downgradient wells did not show similar concentrations (see trace metal attenuation discussion below). In other interior site field studies, trace metal concentrations in waste water violated Primary Drinking Water Standards at 3 of 5 sites (Chisman Creek, Wallingford, Bailly, Zuellinger and Hunts Brook sites; see Reference 13). At one site, selenium bioaccumulation in a combined treatment pond was reported to be responsible for changes in a fish population (73).
- The typical range of interior climatic and hydrogeologic conditions can result in enough leachate generation and movement from unlined disposal sites to have a measurable effect on downgradient groundwater quality. Unlined pond and landfill sites in typical interior settings in this project (Powerton and Allen sites) contributed measurable leachates to downgradient groundwater. This same phenomenon was observed in most other studies of interior sites, regardless of the method of disposal (Wallingford, Bailly, Hunts Brook and Zuellinger site results reported in Reference 13). At Sherburne County, the lined pond site in this project, leachate movement was significantly restricted, at least for the short term.
- Admixing of leachates with even small surface water bodies can have a major dilution effect that mitigates potential leachate impacts on surface water and groundwater quality. Results from this project indicate that adjacent small streams can achieve an almost instant, order-of-magnitude reduction in landfill leachate concentrations of major dissolved species (see Powerton site results in Section 5.2.5). Other interior site studies confirm this finding (see, for example, Hunts Brook site results reported in Reference 13).
- Field data are incomplete as to how much effective dilution may be achieved in surface waters with the larger leachate flows that may result downgradient of interior pond disposal areas. For instance, the large impoundments required for pond disposal of FGD wastes (with their high levels of major dissolved species such as  $\text{Cl}^-$ ) would have more significant drainage effects on water quality in a small, adjacent surface water body than the other disposal methods would.
- Significant chemical attenuation of trace metals in soils can be a mitigating factor in some, but not all prevalent soil types.

Background soils with high clay, iron and manganese contents at one of the three typical interior sites studied in this project appeared to have substantially attenuated As in ash pond leachate (see Allen site results, Section 5.2.1). At another site (Powerton), such As attenuation was not apparent in at least one downgradient location. Selenium attenuation was possible in the liner at the third typical interior site (Sherburne County), but the sandy, inorganic background soils showed little attenuation potential. In two other field studies at sites in the Indiana Lake Plain, chemical attenuation of trace metals in soils was reported (see results for Bailly and Michigan City sites in Reference 13).

Adverse effects of coal ash or FGD waste disposal by various methods in typical interior settings seem limited to situations where unlined sites are developed so close to useful water supplies that elevated leachate concentrations of major and/or minor species can reach the water supply without being diluted by an intervening surface water body. In a few cases, leachates from a large disposal pond or from a site with unusually high trace metal concentrations and non-attenuative soils also may be of concern for a small, adjoining surface water body. If such conditions were expected, the problem could be alleviated by choosing an alternative disposal site, placing an impermeable and/or chemically attenuative soil as a site liner, or by managing the site water to prevent contaminated leachate from reaching the receiving waters.

#### 7.3.8 Summary of Effects Implications of Disposal in Interior Settings Affected by Acid-Mine Drainage; Various Waste Types and Disposal Methods

##### 7.3.8.1 Practices in this Category--

The interior setting affected by acid mine drainage is most frequently encountered in the Central Appalachian Mountains coal-mining region of the Eastern U.S. - in such states as Pennsylvania, West Virginia, Ohio and Kentucky. In some locations, many years of surface and subsurface mining have significantly affected background water quality in the smaller surface water bodies and in mine-associated groundwaters. In other respects (i.e., climate and geohydrology), these areas are similar to the typical interior setting discussed in Section 7.3.7.

Of the possible combinations of waste types, disposal methods and environmental settings shown in Table 7.1, the same waste/disposal combinations applicable to typical interior settings (Section 7.3.7) may also occur in this setting. (The prevalence of these combinations in Central Appalachia is essentially the same as that described in Section 7.3.7.1).

##### 7.3.8.2 Status of Documentation--

A full-scale stabilized FGD waste landfill in an interior acid mine drainage setting in western Pennsylvania was studied as part of this project (see Elrama site results in Section 5.2.2). At least two of the "typical" interior sites studied by others are near documented or suspected mining

activity (72; Conesville site results reported in Reference 13), but these investigations have not identified any acid drainage that may prevail.

Results from the many field studies of the generic effects of interior climate and geohydrology on coal ash and FGD waste disposal (see Section 7.3.7.2) can be combined with the findings of this project, which concerned, in addition to these issues, the unique chemistry of interaction between a highly alkaline, stabilized FGD waste and its acid mine drainage setting. This information provides an adequate basis for a discussion of the environmental implications.

#### 7.3.8.3 Summary of Environmental Implications--

The various coal ash and FGD wastes are likely to have certain generic environmental implications when disposed of by any of the major methods in interior acid mine drainage settings:

- Leachate concentrations of major dissolved species, such as sulfate, have little, if any, potential to degrade significantly receiving waters that are already affected by acid mine drainage. As discussed in Section 7.2, the background water quality in areas subject to acid mine drainage is so poor that the Secondary Drinking Water Standards are chronically exceeded, and surface water bodies are unable to support the full range of aquatic life uses. An important issue to consider is whether the slight addition of leachate contaminants may preempt longer term uses. Results from this project indicate that such preemptions are unlikely.
- In some cases, certain trace metals in coal ash or FGD waste leachate (i.e., As, Se) may be of concern. At Elrama, the acid mine drainage site studied in this project, As concentrations at one leachate collection point immediately below the waste deposit were 3 to 5 times higher than the Interim Primary Drinking Water Standard of 50 µg/l. The highly acidic nature of the background mine drainage could, at some sites, promote leaching of some of the otherwise less available trace metal fractions of coal ash or FGD waste (i.e., Cu, Zn), especially from fly ash. But relevant field-scale data are lacking. It is also possible that leachate from some of the more highly-alkaline ash or FGD wastes can locally reduce (by attenuation) the availability of trace elements that would otherwise remain relatively free in the mine drainage. This may occur at Elrama, although the site has not been operated long enough to be certain.
- As in the typical interior setting, the climatic and hydrogeologic conditions in regions experiencing acid mine drainage can cause leachate generation and movement from unlined disposal sites, with a measurable effect on downgradient groundwater quality. These effects can be measured by following some of the more environmentally innocuous tracer species, such as Ca. But the likelihood of incremental degradation in these waters is far less than in the typical interior settings, where receiving water bodies are more likely to support a full range of actual and potential uses.



- As in typical interior settings, admixing of leachates with even small surface water bodies can have a major dilution effect that mitigates potential leachate impacts on surface water and groundwater quality. This is even more true in acid mine drainage settings, where background and leachate concentrations of most major species are similar.
- Significant chemical attenuation of trace metals in soils can be a mitigating factor in some, but not all of the prevalent circumstances. The acidity of the mine drainage and the wide pH range of various coal ash and FGD waste leachates may also combine to create locally unusual metals attenuation (or release) phenomena. Field-scale documentation is lacking on attenuation phenomena. At the Elrama site studied in this project, the soil attenuation capacity for trace metals was in the high to intermediate range (Appendix F).

Adverse effects of coal ash or FGD waste disposal by various methods in interior acid mine drainage settings with contaminated backgrounds appear to be limited to those cases where leachates from a disposal area with extremely high, leachable trace metal levels could reach and remain undiluted in a small water body used for such purposes as drinking or fishing, in spite of the prevalent acid mine drainage. If these unlikely conditions were expected, a variety of mitigative measures could be taken (see Section 7.3.7.3).

#### 7.4 DATA GAPS

For many of the important combinations of waste types, disposal methods and environmental settings described in Section 7.3, the existing information base provides a solid foundation for estimating the overall potential for adverse effects on water quality and for disposal-related decision-making. But in some areas, significant data gaps exist, as described below.

##### 7.4.1 Characterization of Trace Metals in Waste Leachates.

Although much information has been developed in this area (see Section 3.4), an analytical framework is lacking for accurately anticipating leachate trace metal concentrations based on coal types, air pollution control systems, and disposal methods. The data base remains fairly incomplete for trace metal speciation in leachates, for wastes produced by dry sorbent FGD systems, and for coal ashes collected by bag filters that retain a greater percentage of the trace-metal-enriched fine particles than other particulate control systems. Since leachate trace metal composition is an important decision point, this is a high priority area for research. EPRI is investigating this area now.

##### 7.4.2 Soil Attenuation of Leachate Constituents

Data developed during this project confirm the basic premise that several major factors, including pH, soil organic content, soil particle size, and ion exchange tendencies, combine to make prospective soil attenuation of leachate trace metals a highly variable, site-specific phenomenon. Soil attenuation

can significantly reduce the potential for adverse effects at a disposal site (see the results for the Allen site in Section 5.2.1), and a more complete data base would be valuable for projecting impacts. Such attenuation issues are being addressed in studies sponsored by EPRI.

#### 7.4.3 Effects of Various Artificial Liners on Long-Term Leachate Movement and Composition

Some of the artificial liners (made of material imported from off-site) studied in this project (i.e., soils at Sherburne County) can greatly reduce the rate of leachate release from a disposal area (see Section 5.2.4). This ability to retain water can be readily predicted. But it is difficult to project the impact of such liners on subsequent long-term leachate movement and admixing in adjoining unsaturated zones. Chemically attenuative soils can be used as liner materials, and these may serve as a practical mitigative measure in certain settings. The development of a regional data base correlating the relative abilities of potential liner materials to retard water movement and attenuate selected trace metals would be of obvious value.

#### 7.4.4 Pond Versus Landfill Disposal In Arid Settings

Field studies in arid settings have focused on landfill disposal operations. Although landfill disposal will likely be the dominant practice for new operations in the western U.S., pond disposal will also persist. A better field-scale data base is needed to clarify how the maintenance of hydraulic head by ponding affects leachate movement and the site environment. This information would be especially valuable in arid, non-highly mineralized settings, where waste leachate quality differs significantly from that of the receiving water bodies.

#### 7.4.5 Impacts of Abnormal Events

The existing field-scale data base largely describes cumulative effects observed at various study sites. As such, it reflects normal operations versus abnormal events (storms, earthquakes, material/equipment failure, etc.). For some circumstances, the data suggest that abnormal events can have proportionately much greater short-term impacts on groundwater quality than the normal disposal operation (see Section 5.2.4). In certain combinations of waste, disposal method and setting, the likelihood of abnormal events may warrant inclusion in the disposal-related decision-making process. For example, the potential for major storm events could be an important design consideration in a coastal ponding operation.

#### 7.4.6 Effects of Acid Mine Drainage and Waste Alkalinity on Trace Metal Release

Results of this project show how acid mine drainage conditions can interact with a highly-alkaline, processed FGD waste. The disposal of less alkaline (and even slightly acidic) coal ashes in acid mine drainage situations may possibly increase the release of trace metals in leachate. Very high waste alkalinity may mobilize some trace metals in some other disposal



settings (68). Further field-scale data on such phenomena would greatly aid the waste disposal decision-making process.

#### 7.4.7 Admixing of Pond Leachates with Small Surface Water Bodies

Field-scale data strongly suggest that even very small permanent surface water bodies can cause major mitigative dilution impacts upon admixing with leachates from coal ash or FGD waste landfills. But data are lacking as to how such interaction would occur with the significantly greater leachate volumes generated by pond disposal operations, especially those involving very large water volumes and relatively high concentrations of dissolved major species (i.e., FGD waste ponds). Field scale studies to clarify this area would be practical at existing disposal sites. Data from such studies would be valuable in waste disposal decision-making, where location relative to a surface water body is often a major consideration.

## SECTION 8.0

### DECISION METHODOLOGY

#### 8.1 INTRODUCTION

This Section provides various tools to assist utility planners and permitting officials in evaluating waste management practices:

- A decision methodology that outlines a procedure to help answer questions about coal ash and FGD waste disposal.
- A list of information requirements for assessing the waste management potential alternatives, including recommended laboratory and field tests that may provide more detailed information on a specific site.
- An information base on environmental effects and engineering costs associated with various waste management options. This gives some initial guidance on how to evaluate waste characteristics, disposal methods, and environmental settings and select the proper waste management alternative.

These tools should serve as guidelines only. Ultimate environmental effects of coal ash and FGD waste disposal are highly site- and system-specific and cannot be generalized. Table 8.1 shows the general environmental impact issues associated with various disposal options. Appropriate control technology can be applied on a site-specific basis to mitigate adverse impacts. The significance of many potential impacts may be better quantified by additional laboratory or field-scale work.

#### 8.2 WASTE MANAGEMENT ISSUES

A state or local permitting official or a utility planner may face several waste management issues related to coal ash and FGD wastes. Typical examples are:

- Various waste management options for new coal-fired power plants.
- Waste management options resulting from expansion of existing power plants or changes in disposal operations. A typical example would be adding an FGD system that will generate FGD waste. Another would be a change in the disposal site. One ash landfill might have been filled, and another must be brought into operation.

TABLE 3.1

## POTENTIAL IMPACT ISSUES--COAL ASH AND FGD WASTE DISPOSAL

| Setting                                               | Disposal Practice                    |                       |
|-------------------------------------------------------|--------------------------------------|-----------------------|
|                                                       | Ponding                              | Interim Pond/Landfill |
| Coastal                                               | G-2, S-2, 3, M-2                     | G-2, S-2, 3, M-2      |
| Interior                                              | G-2, S-2, M-2                        | G-2, S-2, M-2         |
| Interior - Acid Mine Drainage                         | G-2, S-2, 3, M-2                     | G-2, S-2, 3, M-2      |
| Arid                                                  | G-2, S-2, M-2                        | G-2, S-2, M-2         |
| Arid Mineralized                                      | G-3, S-3, M-3                        | G-3, S-3, M-3         |
| Type of Issue                                         | Level of Importance                  |                       |
| C = Effects on groundwater quality.                   | 1. Usually important.                |                       |
| S = Non-point effects on surface water quality.       | 2. Important in site-specific cases. |                       |
| M = Mitigative design, management or control practice | 3. Usually of minor importance.      |                       |

Note: For details on settings see Section 7.

- Waste management alternatives associated with power plant conversion to coal from some other form of fossil fuel.
- The affects of continuing existing operations over the next few years.
- Impact of abnormal events such as a hurricane or a flood on a waste disposal site.

Each issue requires a different perspective and approach in the initial evaluation of alternatives. Some may involve making projections into the future from an existing data base. But, whatever the various issues, the overall environmental effects of coal ash and FGD waste disposal can be determined, given information on waste characteristics disposal methods (including engineering and cost considerations), and environmental settings. (Naturally, the more complete the data base the more reliable the assessment will be.) This initial assessment will enable some preliminary alternatives to be identified. After additional refinement by laboratory and field tests, more suitable waste management options can be selected.

### 8.3 DECISION METHODOLOGY

This Section describes the generic decision methodology developed to help the decision maker arrive at proper waste management options. The steps involved in evaluating alternatives for coal ash and FGD waste disposal are briefly outlined in Figure 8.1. The sequence is:

- Step 1 - Gather Data and Information on Disposal Alternatives

A starting point for evaluating the options is to develop a list of information requirements for the waste and the type of disposal that may be feasible. These information requirements are discussed in Section 8.4. At this stage in the decision-making process, the approach is to gather as much data as possible on the potential type of wastes and the various types of disposal options.

- Step 2 - Perform Preliminary Engineering Evaluation of Disposal Alternatives

After the initial information requirements have been met, a preliminary list of the potentially applicable disposal options should be developed. This list should be as complete as possible, based on an evaluation of the detailed data base provided by this report (Section 6) and the references cited. The data base includes waste characteristics and disposal methods and associated capital and annual costs.

- Step 3 - Gather Data and Information on Environmental Settings

This step involves defining the information requirements on the site-specific environmental setting, including geology, hydrogeologic conditions, surface water information, climatic conditions, and other

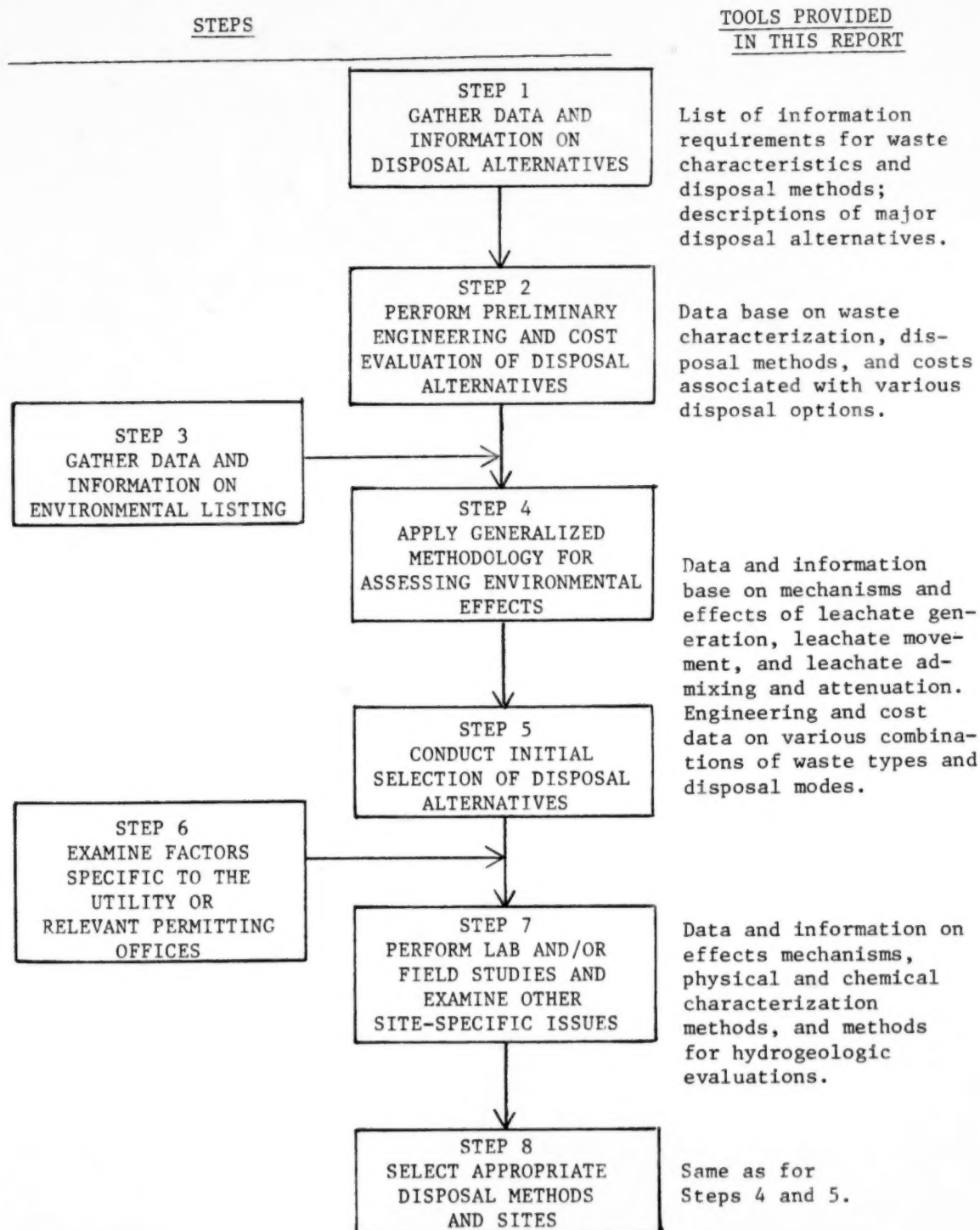


FIGURE 8.1 OVERALL DECISION-MAKING PROCESS

aspects of the environment. Section 8.4 describes the typical information requirements. Appendices to this report, together with the discussions in Section 5 and 6, provide the data base needed to compile this information.

- Step 4 - Apply Generalized Methodology for Assessing Environmental Effects

Once information is available on waste characteristics, methods of disposal, and environmental settings, the next step is to evaluate the environmental effects. This can be best undertaken by studying the processes leading to leachate generation, leachate movement, leachate admixing with receiving water, and chemical attenuation by soils. Each of these four subjects is covered in detail in Section 8.6. The engineering and cost information on various combinations of different waste types and disposal modes, provided in Section 6 of this report and in the appendices, supplement the data base. This information allows both the environmental effects of each type of disposal and the associated conceptual costs to be evaluated.

- Step 5 - Make Initial Selection of Disposal Alternatives

From a compilation of the environmental effects of various alternatives and the engineering costs associated with each, the decision-maker can perform a comparative assessment to arrive at an initial list of alternatives that may require the additional scrutiny described below.

- Step 6 - Examine Factors Specific to the Utility or Relevant Permitting Offices

Each utility may have some historical practice in terms of waste management. Similarly, there may be very specific policy requirements that the permitting offices have to fulfill. These specific items are not covered in detail in this report. But the list of references provided cover the issues involved and provide an information base to assist the decision-maker.

- Step 7 - Perform Lab and/or Field Studies and Examine Other Site-Specific Issues

Once a preliminary list of disposal options is available, laboratory or field tests may be necessary to obtain additional information on waste characteristics (both physical and chemical) or on the environmental settings. More site-specific engineering cost evaluations may also be in order. This report provides detailed information on physical and chemical characterization methods (see Appendix C), hydrogeologic evaluations (see Appendix B), and other procedures involved in determining environmental effects. Also, Section 6 provides engineering cost information for more site-specific disposal systems.



## • Step 8 - Select Appropriate Disposal Methods and Sites

From the information developed under Step 7 and the assessment from Step 5, the decision-maker can arrive at the most appropriate disposal-site and method for the specific situation.

These decision-making steps will vary from one situation to another. In some cases, substantial laboratory data and field effort may be required; in others, Step 5 may provide a clear enough indication of the waste management options to address the issues involved. The references given in Sections 6, 7, and 8 provide the additional information required for more detailed evaluation.

### 8.4 INFORMATION REQUIREMENTS

The information needed to consider environmental effects concerns waste type, disposal practice, and setting (see Figure 8.2). Specific requirements are listed in Tables 8.2 (waste type), 8.3 (environmental setting) and 4.11 (disposal method, see Section 4.6). The information requirements shown in these tables are quite complete. In many instances, all the information may not be available or may not be required. If necessary, data gaps should be filled. Laboratory or field efforts to obtain more specific data may be called for (see Appendixes B & C and References 2, 3, 17, 21, 60).

### 8.5 ENGINEERING/COST CONSIDERATIONS

The engineering data and cost estimation methodology provided in Section 6 of this report can be applied as described in Step 5 of the Decision Methodology to define and select appropriate waste handling/disposal scenarios for the new coal-fired power plant under consideration. Conceptual capital and annual cost estimates for the scenarios can then be selected. This process is described briefly below. See Section 6 for more detailed information.

Once the preliminary engineering design premises for the newly proposed plant have been established, they must be compared to those design premises on which the costs were based (see Table 6.5, Section 6). Any differences between the two must be noted so that appropriate correction factors can be determined. A list of the potentially applicable disposal options is then developed. This list should be as complete as possible, based on an evaluation of the detailed engineering data base provided in Section 6.1 and the numerous references cited.

Variations among waste types and in the collection, handling, processing, storage, transport and disposal of these wastes call for different approaches to waste management. Consequently, a matrix of waste types and waste management activities has been developed. For each type/waste management activity combination, one or more design option has been presented. The desired modular options can be combined to develop a preliminary engineering specification which is then used in determining a conceptual cost estimate for the desired integrated waste management system. An engineering description of

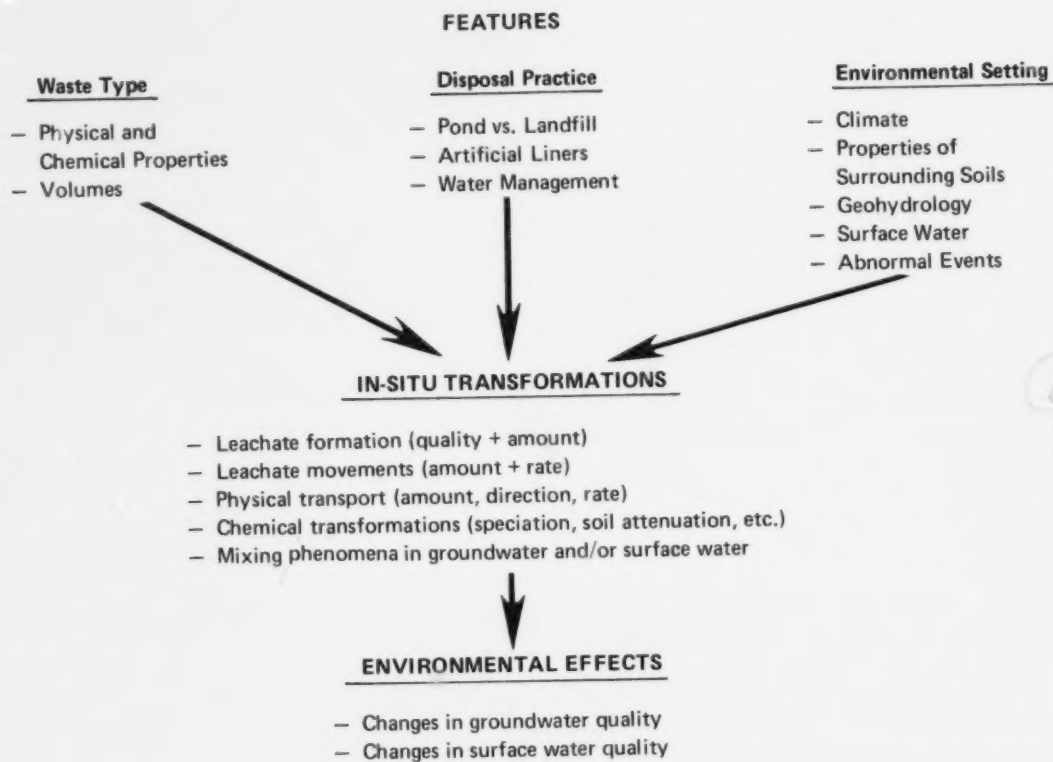


FIGURE 8.2 MAJOR FEATURES OF THE ENVIRONMENTAL EFFECTS PROCESS

TABLE 8.2

POTENTIAL INFORMATION REQUIREMENTS ON WASTE TYPE<sup>a</sup>

1. Power Plant Size and Location
2. Waste Type: Fly Ash  
Bottom Ash  
FGD, Waste, etc.
3. Volume: Annual Average  
Maximum in any Given Period  
Minimum in any Given Period

## 4. Physical Properties:

| <u>Average</u> | <u>Maximum</u> | <u>Minimum</u> |
|----------------|----------------|----------------|
|----------------|----------------|----------------|

## a. Coal Ash

## (1) Grain Properties

Specific Gravity  
Grain Size  
Coefficient of Uniformity  
Atterburg Limits

## (2) Compaction Properties

Bulk Dry Density  
Field Density  
Controlled Compacted Density

## (3) Permeability

## (4) Strength Parameters

## b. FGD Wastes

## (1) Grain Properties

Specific Gravity  
Grain Size

## (2) Coefficient of Uniformity

Atterburg Limits

(continued)

TABLE 8.2

|                                            | <u>Average</u> | <u>Maximum</u> | <u>Minimum</u> |
|--------------------------------------------|----------------|----------------|----------------|
| (3) Compaction Properties                  |                |                |                |
| Maximum Dry Density                        |                |                |                |
| Optimum Moisture Content                   |                |                |                |
| Compressibility                            |                |                |                |
| (4) Permeability                           |                |                |                |
| (5) Strength Parameters                    |                |                |                |
| Angle of Internal Friction                 |                |                |                |
| Effective Cohesion                         |                |                |                |
| 5. Chemical Properties                     |                |                |                |
| a. Coal Ash Wastes, Liquors and Elutriates |                |                |                |
| (1) Major Constituents                     |                |                |                |
| Silicon Dioxide                            |                |                |                |
| Aluminum Oxide                             |                |                |                |
| Iron Oxide                                 |                |                |                |
| Calcium Oxide                              |                |                |                |
| Sodium Oxide                               |                |                |                |
| Magnesium Oxide                            |                |                |                |
| (2) Minor Constituents                     |                |                |                |
| Titanium Oxide                             |                |                |                |
| Sulfur Oxide                               |                |                |                |
| Potassium Oxide                            |                |                |                |
| Phosphorous Pentoxide                      |                |                |                |
| (3) Trace Constituents                     |                |                |                |
| Arsenic                                    |                |                |                |
| Beryllium                                  |                |                |                |
| Boron                                      |                |                |                |
| Cadmium                                    |                |                |                |
| Chromium                                   |                |                |                |
| Copper                                     |                |                |                |
| Lead                                       |                |                |                |
| Manganese                                  |                |                |                |
| Mercury                                    |                |                |                |
| Nickel                                     |                |                |                |

(continued)

TABLE 8.2

|                                      | <u>Average</u> | <u>Maximum</u> | <u>Minimum</u> |
|--------------------------------------|----------------|----------------|----------------|
| (3) Trace Constituents (cont.)       |                |                |                |
| Selenium                             |                |                |                |
| Strontium                            |                |                |                |
| Vanadium                             |                |                |                |
| Zinc                                 |                |                |                |
| b. FGD Wastes Liquors and Elutriates |                |                |                |
| (1) Major Constituents               |                |                |                |
| Calcium                              |                |                |                |
| Chloride                             |                |                |                |
| Magnesium                            |                |                |                |
| Potassium                            |                |                |                |
| Sodium                               |                |                |                |
| Sulfate                              |                |                |                |
| Fluoride                             |                |                |                |
| Total Dissolved Solids               |                |                |                |
| pH                                   |                |                |                |
| (2) Trace Constituents               |                |                |                |
| Antimony                             |                |                |                |
| Arsenic                              |                |                |                |
| Beryllium                            |                |                |                |
| Boron                                |                |                |                |
| Cadmium                              |                |                |                |
| Chromium                             |                |                |                |
| Cobalt                               |                |                |                |
| Copper                               |                |                |                |
| Iron                                 |                |                |                |
| Lead                                 |                |                |                |
| Manganese                            |                |                |                |
| Mercury                              |                |                |                |
| Nickel                               |                |                |                |
| Selenium                             |                |                |                |
| Strontium                            |                |                |                |
| Vanadium                             |                |                |                |
| Zinc                                 |                |                |                |

Note: <sup>a</sup>List shown is comprehensive; all requirements may not be necessary in a given decision-making process.

Source: Arthur D. Little, Inc.

TABLE 8.3

TYPICAL INFORMATION REQUIREMENTS - ENVIRONMENTAL SETTING<sup>a</sup>

1. Power Plant and Location
2. Geotechnical Baseline
  - a. Foundation Conditions Including Cross Sections and Evaluations
  - b. Disposal Site Design and Construction (engineering details)
  - c. Disposal Site Distress
3. Hydrogeologic Baseline
  - a. Geologic Conditions
    - Near-Region Geology
    - Site Geology
    - General Subsurface Profile
  - b. Hydrogeologic Conditions
    - Surface Water Regime
    - Groundwater Regime
    - Adjacent Water Usage
    - Background Groundwater and Surface Water Quality
  - c. Climatic Conditions
    - Temperature
    - Rainfall
    - Evaporation
4. Other Information
  - a. Area Studies
  - b. Site Studies
    - Land use and zoning information
    - Other, as appropriate (e.g., critical habitat)

Note: <sup>a</sup>List shown is comprehensive; all requirements may not be necessary in a given decision-making process.

Source: Arthur D. Little, Inc.



the modules is presented in Section 6.1.2 to assist in determining which waste handling/disposal system might be best for the specific application.

After the waste handling and disposal alternatives have been selected, the generic cost estimation methodology provided in Section 6.2 can be used as a basis for developing conceptual cost estimates for these alternatives. A set of cost curves is provided as an easy and accurate method for developing capital and annual cost estimates for the waste handling and disposal system of a proposed new coal-fired utility plant. These cost curves are structured to allow estimates of both the capital and annular modular costs for the handling, processing, storage, transport and ultimate disposal of any or all of the three major wastes (i.e., fly ash, bottom ash, FGD waste) generated at coal-fired power plants. In this way, for a given combination of utility plant size (i.e., electric generating capacity) and waste types, the cost curves can be used to estimate the capital and annual operating costs (on a late-1982 basis) for the entire waste handling and disposal system or for any of the individual corresponding process modules comprising that system. Once base capital and annual costs for all of the appropriate waste type/process module combinations have been determined, corrected capital and annual costs (when required) can be calculated.

## 8.6 MECHANISMS OF ENVIRONMENTAL EFFECTS

### 8.6.1 Integration of Effects in Decision-Making

Figure 8.3 shows the type of decision logic that can be used to evaluate the main mechanisms of environmental effects discussed in the rest of Section 8.6. The first decision point is to determine whether the potential disposal site locations are known. If so, a fairly straightforward series of preliminary evaluations may allow the full range of disposal options to be considered without further major concern over environmental (water quality) effects. This series of evaluations would include:

1. Preliminary assessment of leachate formation, movement and admixing. Available information can be used to make a "first-cut" water balance analysis. This will give relatively gross estimates of potential leachate contributions to downgradient receiving water. If the preferred disposal method is known, it should be used for the estimate. If not, a "worst case" analysis should be taken, assuming the candidate method that allows the greatest amount of relatively unconfined water movement (i.e., disposal in an unlined pond). In the same manner, leachate and background water quality should be estimated based on the best available data. Where uncertainty is involved, the worst reasonably expected leachate quality and the best (least contaminated) reasonably expected background water quality should be assumed.
2. Initial judgement of the potential for adverse water quality effects. At this point it may be clear that the disposal operation will be acceptable. The site may be physically removed enough that dilution or attenuation by the soil can mitigate the potential for significant

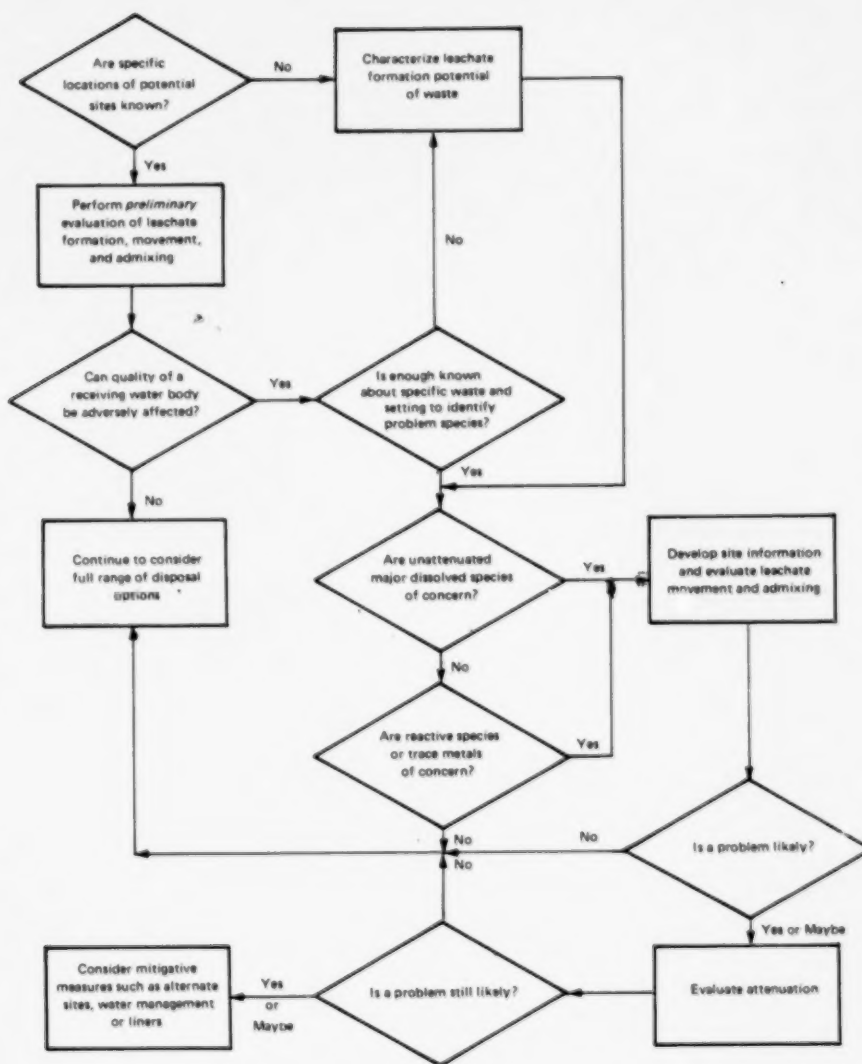


FIGURE 8.3 DECISION LOGIC FOR INTEGRATING EFFECTS

adverse effects. Alternatively, the background water quality may be so poor (i.e., unfit for human consumption) that the leachate would not have significant incremental adverse effects on downgradient water quality. Such conditions might be anticipated in some of the settings described in Section 7.3, especially where arid, coastal or acid mine drainage conditions exist.

If concern or uncertainty remains after these initial considerations, or if the prospective disposal sites are unknown, more detailed analysis is required:

3. Characterization of potentially problematic amounts and concentrations of chemical species in the waste leachate. Depending on how much information is available, this step may involve several of the methods to characterize leachate formation (described in Section 8.6.2). Subsequent decisions will depend on which species are of primary concern - the unattenuated, major dissolved species present in all types of coal ash or FGD waste leachates, or the less frequently problematic trace metal species. Leachate characterization may indicate that, after admixing with background waters, the potential for adverse effects is eliminated.
4. Development of site geohydrologic information and evaluation of leachate movement and admixing. Methods for evaluating leachate movement and admixing are discussed in Sections 8.6.2 and 8.6.3. These evaluations will indicate whether unacceptable downgradient concentrations of one or more leachate species are likely. If a reactive species or trace metals-related problem is indicated, the potential for soil attenuation should be assessed, as explained in Section 8.6.4.
5. Consideration of mitigative measures if the problem still seems likely. Potentially mitigative practices (alternate disposal sites, water management practices, and site liners) are discussed in Section 6. Details on engineering/cost considerations are presented in Section 6.2.

#### 8.6.2 Leachate Generation: Movement and Chemical Composition

##### 8.6.2.1 Leachate Movement--

Leachate movement is a complicated phenomenon affected by waste disposal method, climatological conditions and hydrogeology. The method of waste disposal may be either landfill, where the background water balance for the site is of primary concern, or ponding, where hydraulic head conditions in the pond and hydraulic conductivity of underlying materials are most important. Climatological conditions may range from arid, characterized by intense intermittent precipitation events and potential evapotranspiration (EVTs) exceeding precipitation, to humid, typified by year-round precipitation and potential EVT's less than precipitation. The regional hydrogeology of a waste disposal facility is defined by many parameters - the presence and stratigraphy of aquifers, depth to water table and thickness of the

unsaturated zone, vertical and horizontal hydraulic conductivities and gradients, directions and rates of groundwater movement, presence and hydraulic properties of bedrock, groundwater-surface water interactions, and man-made groundwater discharges such as wells.

#### 8.6.2.1.1 Disposal Method--

Landfill -- Determining leachate movement from a landfill involves estimating the quantity of water that infiltrates into and through the landfill. Figure 8.4 shows the flow of water in a landfill disposal site. The four primary sources of water in a landfill are:

- process water in the landfilled waste,
- percolation of direct precipitation,
- percolation of runoff onto the landfill, and
- groundwater movement through the landfill if it is below the water table.

The extent of water in the waste depends on waste handling and disposal processes. This water is usually in small amounts - insufficient to saturate the waste. Percolation of direct precipitation and surface water is related to the physical properties of the landfill (permeability, waste moisture capacity) and local climatological conditions. The amount of percolation can be estimated by the equation:

$$\text{PERC} = I - \Delta\text{ST} - \text{AET} \quad (1)$$

where

PERC = percolation  
I = infiltration  
 $\Delta\text{ST}$  = change in waste moisture storage  
AET = actual evapotranspiration

The amount of water which enters the landfill surface, or infiltration, is described by the equation:

$$I = P + R_I - R_O \quad (2)$$

where

I = infiltration  
P = precipitation  
 $R_I$  = runoff onto landfill  
 $R_O$  = runoff from landfill

This represents the maximum amount of water available to move through the landfill.

Precipitation varies with location. Historical records of precipitation are available from various state and federal agencies, most notably the

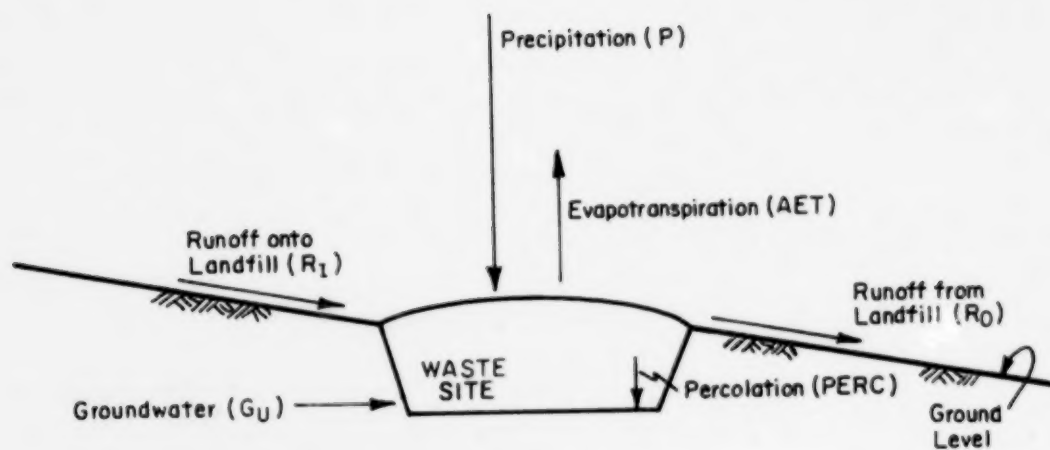


FIGURE 8-4 WATER FLOW IN A LANDFILL DISPOSAL SITE.

National Oceanic and Atmospheric Administration (NOAA). Runoff onto the landfill is a function of topographic and drainage conditions adjacent to the landfill. Runoff can be minimized by using diversion ditches around the site. The amount of infiltrating water is depleted by changes in soil moisture storage, direct evaporation to the atmosphere, and transpiration by plants. The remaining water moves through the surface layer, percolates through the landfill, and will eventually emerge as leachate.

An additional source of water in landfills is groundwater flow through the landfill, or underflow. This occurs if the base of the landfill is below the water table. In this case, the amount of leachate can be estimated by the equation:

$$L = \text{PERC} + G_u \quad (3)$$

where

- L = amount of leachate
- PERC = percolation through landfill
- $G_u$  = groundwater underflow through landfill

The amount of groundwater underflow can be estimated from water table conditions in the vicinity of the landfill.

Ponds -- In a pond, standing water provides the driving force for moving leachate out of the pond and into adjacent materials. Since most of this movement is through the bottom of the pond, the predominant driving forces and direction of movement vertical. The vertical movement of leachate can be impeded by underlying man-made or natural less permeable materials. Figure 8.5 shows the flow of water in a pond disposal site.

The movement of leachate from a pond may be described in terms of hydraulic potential or mass balance. The hydraulic potential approach is described by a form of Darcy's Law, which considers the driving forces or hydraulic head in the pond, the hydraulic conductivity of waste and underlying materials, and the area of the pond:

$$Q = KiA \quad (4)$$

where

- Q = water movement through pond bottom, or seepage
- K = hydraulic conductivity of underlying material
- i = hydraulic gradient across underlying material
- A = area of pond bottom

Equation 4 applies to ponds where the area of the bottom is much greater than the area of the sides, and where most of the seepage occurs through the pond bottom. The controlling terms in the equation are K and i. K is the hydraulic conductivity of the underlying material. This may be a man-made liner or a natural soil or rock unit. If waste material has settled on the pond bottom, it is considered part of the effective underlying material. The



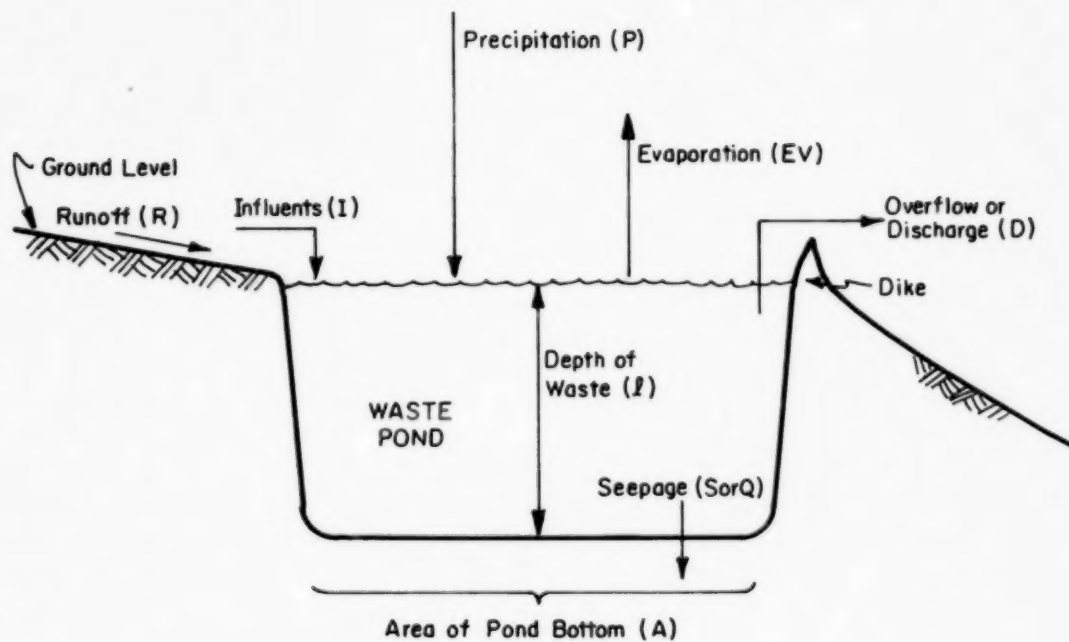


FIGURE 8-5 WATER FLOW IN A POND DISPOSAL SITE.

hydraulic gradient is a measure of the hydraulic head distribution across the underlying material and is given by the equation:

$$i = \Delta h / l \quad (5)$$

where

$\Delta h$  = head difference across the material

$l$  = length or thickness of material

The head difference is equal to the difference between the hydraulic head at the bottom of the pond (usually the water level in the pond) and the hydraulic head at the base of the underlying material. This may be equal to or less than one atmosphere if the underlying soil or rock is unsaturated, or greater than one atmosphere if it is saturated.

The mass balance approach considers the balance of water inputs to and discharges from the pond:

$$S = P + R + I - EV - D \quad (6)$$

where

$S$  = seepage from pond bottom

$P$  = direct precipitation into pond

$R$  = runoff into pond

$I$  = other water inputs to pond (for example, from a discharge line)

$EV$  = direct evaporation from pond

$D$  = other water discharges from the pond (for example, from a spillway or discharge pipe)

Precipitation into the pond may be determined from climatologic records, as may evaporation. More detailed information can be obtained from precipitation gauges and evaporation pans established at the pond site itself. Other water inputs and discharges can be determined from records of the pond operation.

8.6.2.1.2 Hydrogeology -- Once leachate enters the surrounding environment from landfills or ponds, its movement is controlled by hydrogeological conditions in the vicinity. Since leachate will generally move with the groundwater, the direction of its movement can be estimated by studying groundwater flow conditions. Three aspects of site hydrogeology control leachate movement: depth of water table and thickness of the unsaturated zone, hydraulic conductivity anisotropies, and regional groundwater movement.

Thickness of Unsaturated Zone -- Unless the water table is at the ground surface, an unsaturated zone will exist above groundwater. Deep water tables are characterized by thick unsaturated zones while shallow water tables are characterized by thin unsaturated zones. Construction of a pond or landfill will likely alter the geometry of the unsaturated zone unless a completely impermeable (synthetic) liner is placed under the site. Landfill construction may also affect the geometry of the unsaturated zone, as described below.

In a pond or landfill constructed above the water table, initial leachate movement will be vertically downward through the unsaturated zone (see Figure 8.6). If this movement is significant, the water table will mound to increase local hydraulic gradients and cause more movement of groundwater away from the mound (Figure 8.6a). The mound will reach equilibrium when the hydraulic gradients are sufficient to move all the water supplied by the waste disposal facility. If the unsaturated zone is thick, mounding should not raise the water table to the base of the pond or landfill. If the unsaturated zone is thin, mounding may cause direct contact between waste and groundwater and possibly increase rates of leachate generation. The amount of groundwater mounding can be calculated with the governing differential equations.

Under certain conditions, a wetting front may form beneath a pond or landfill and eventually reach the water table (Figure 8.6b). This happens when the saturated hydraulic conductivity of underlying material is less than the percolation or seepage rate. In this case, water is supplied to the underlying material faster than it can move through it. The seepage or percolation is "backed up," and a wetting front formed. The movement of a wetting front to the water table is halted by changes in hydraulic properties of underlying materials, and so would only be expected in very uniform deposits.

Hydraulic Conductivity Anisotropies -- Stratification of soils and sedimentary rock units and structural discontinuities in bedrock units will result in anisotropy of hydraulic properties, most notably hydraulic conductivity. In soils and sedimentary rocks, the horizontal hydraulic conductivity ( $K_h$ ) is usually greater than the vertical hydraulic conductivity ( $K_v$ ). In bedrock, hydraulic conductivity is largest in the directions of structural discontinuities such as joints, faults, and solution cavities.

In a stratified sequence of soil or rock, the equivalent vertical hydraulic conductivity of the section is given by the equation:

$$K_v = \frac{n}{\sum_{i=1}^n} \frac{d_i}{K_{iv}} \quad (7)$$

where

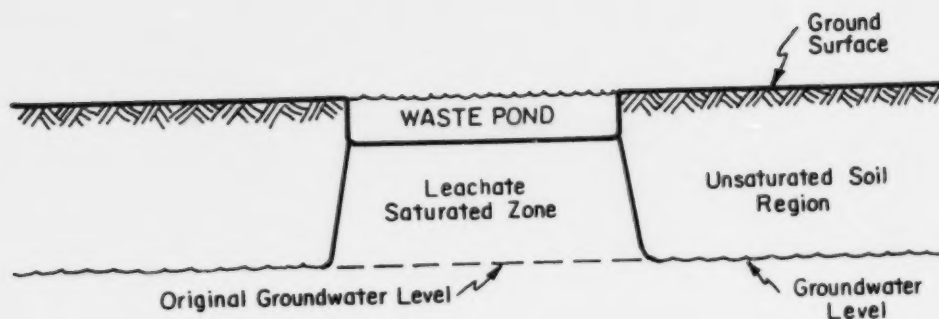
$K_v$  = equivalent vertical hydraulic conductivity

$d_v$  = total section thickness

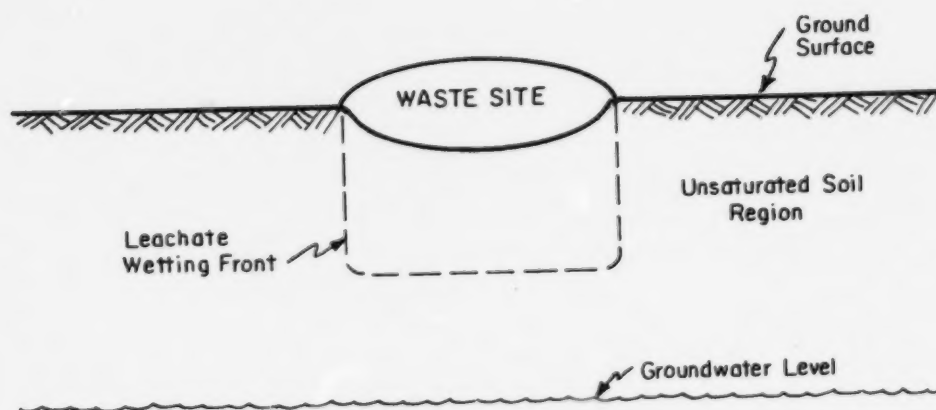
$d_i$  = thickness of individual stratum

$K_{iv}$  = vertical hydraulic conductivity of individual stratum

Equation 8.7 shows that layers with low vertical hydraulic conductivities have the greatest effect on the equivalent hydraulic conductivity of the section. One or two layers with very low hydraulic conductivity may be enough to make the equivalent vertical hydraulic conductivity very low.



(a) Thin unsaturated zone.



(b) Thick unsaturated zone.

FIGURE 8-6 LEACHATE MOVEMENT IN THE UNSATURATED SOIL REGION

The equivalent horizontal hydraulic conductivity of a stratified sequence is given by the equation:

$$K_h = \frac{n}{\sum_{i=1}^n} \frac{K_{ih} d_i}{d} \quad (8)$$

where

- $K_h$  = equivalent horizontal hydraulic conductivity
- $d$  = total section thickness
- $d_i$  = thickness of individual stratum
- $K_{ih}$  = horizontal hydraulic conductivity of individual stratum

The greatest influence on equivalent horizontal hydraulic conductivity will thus be from strata with high horizontal conductivities. One or two strata with high horizontal hydraulic conductivity may be enough to make the equivalent horizontal hydraulic conductivity high.

Comparison of equations 7 and 8 indicates that  $K_h$  is greater than  $K_v$  for all values of  $K_v$ . Ratios of  $K_h$  to  $K_v$  may be on the order of ten, one hundred, or greater.

In deep groundwater areas with large groundwater mounds beneath the waste disposal facility, leachate movement can be predominantly vertical. The rate of leachate movement controlled by the equivalent vertical hydraulic conductivities in the underlying strata. A clay liner may effectively retard leachate movement from the disposal facility. If the water table is shallow or the landfill or pond is constructed below the water table, the predominant direction of leachate movement is horizontal and controlled by the equivalent horizontal hydraulic conductivity. A permeable stratum between layers of much less permeable materials will facilitate horizontal movement of leachate.

Regional Groundwater Movement -- Groundwater generally flows from recharge areas to discharge areas. Recharge areas may consist of outcrops or subcrops of permeable bedrock units, permeable upland soil units and river valley alluvium in humid climates, and alluvial fans and terrace deposits in arid environments. Discharge areas may comprise springs, streams, rivers, ponds, lakes, bays, or pumping wells.

Depending on the rates of percolation and seepage, landfills and ponds may act as local recharge areas. Formation of a groundwater mound indicates significant groundwater recharge. Waste disposal facilities near streams, ponds, and lakes may be located in areas of groundwater discharge, in which case leachate may have a short pathway to the discharge point. Ponds and landfills in low-lying coastal areas may be recharge areas or may be located in discharge areas, depending on local topographic and hydrogeologic conditions.

Pumping wells act as groundwater discharge points which may enhance natural discharge or create new discharge. Groundwater in the area of influence of the well moves towards the well and, if the area intersects a

pond or landfill, leachate also may move towards the well. Pumping from a well may be sufficient to reverse local groundwater flow directions.

8.6.2.1.3 Leachate Quantity Decision-Making Process -- The leachate quantity decision-making process is shown in Figure 8.7. The goal is to estimate the quantity of leachate that may be produced over a certain period of time. This in turn can be related to the flow or leachate generation-per-unit time. Since the decision methodology is different for a pond versus a landfill disposal scenario, it has two separate branches. In the case of a landfill, leachate generation is produced by infiltration of water (precipitation, runoff on the site, or even intermittent disposal of liquid wastes) and contact of groundwater with the waste.

For the ponding disposal scenario, leachate volume may be calculated by: (1) using a mass balance around the pond (influent vs nonleachate effluents) and determining the leachate by difference and (2) using hydraulic conductivity and other site specific parameters to calculate the flow and thus quantity. The decision-making process in Figure 8.7 uses the mass balance approach and may require further calculations based on the second approach, depending on the available data and accuracy of desired result.

#### 8.6.2.2 Leachate Chemical Composition--

8.6.2.2.1 Overview -- Leachate chemical composition depends on the chemical and physical properties of the waste and how they interact:

- displacement of interstitial liquor present in saturated waste;
- mixing of influent water with displaced interstitial liquor (for unsaturated waste);
- dissolution of readily available species in the waste;
- dissolution of bulk matrix species and associated trapped components;
- solubility and adsorption constraints on concentrations in the leachate;
- concentrations determined by total availability of material from the waste; and
- chemical speciation in wastes and leachates.

8.6.2.2.2 Displacement of Interstitial Liquor -- The chemical composition of leachates from saturated wastes is best estimated from the makeup of the interstitial liquid phase. Wells placed in the saturated waste near the bottom of the disposal areas show the composition of water likely to exit the disposal area. This composition may not only represent the waste present at that particular depth but may also reflect how the percolating liquid has contacted many layers of waste. Since waste layers tend to be inhomogenous (i.e., particle size segregation in a pond), different leaching behaviors are expected for various strata. Concentrations in interstitial



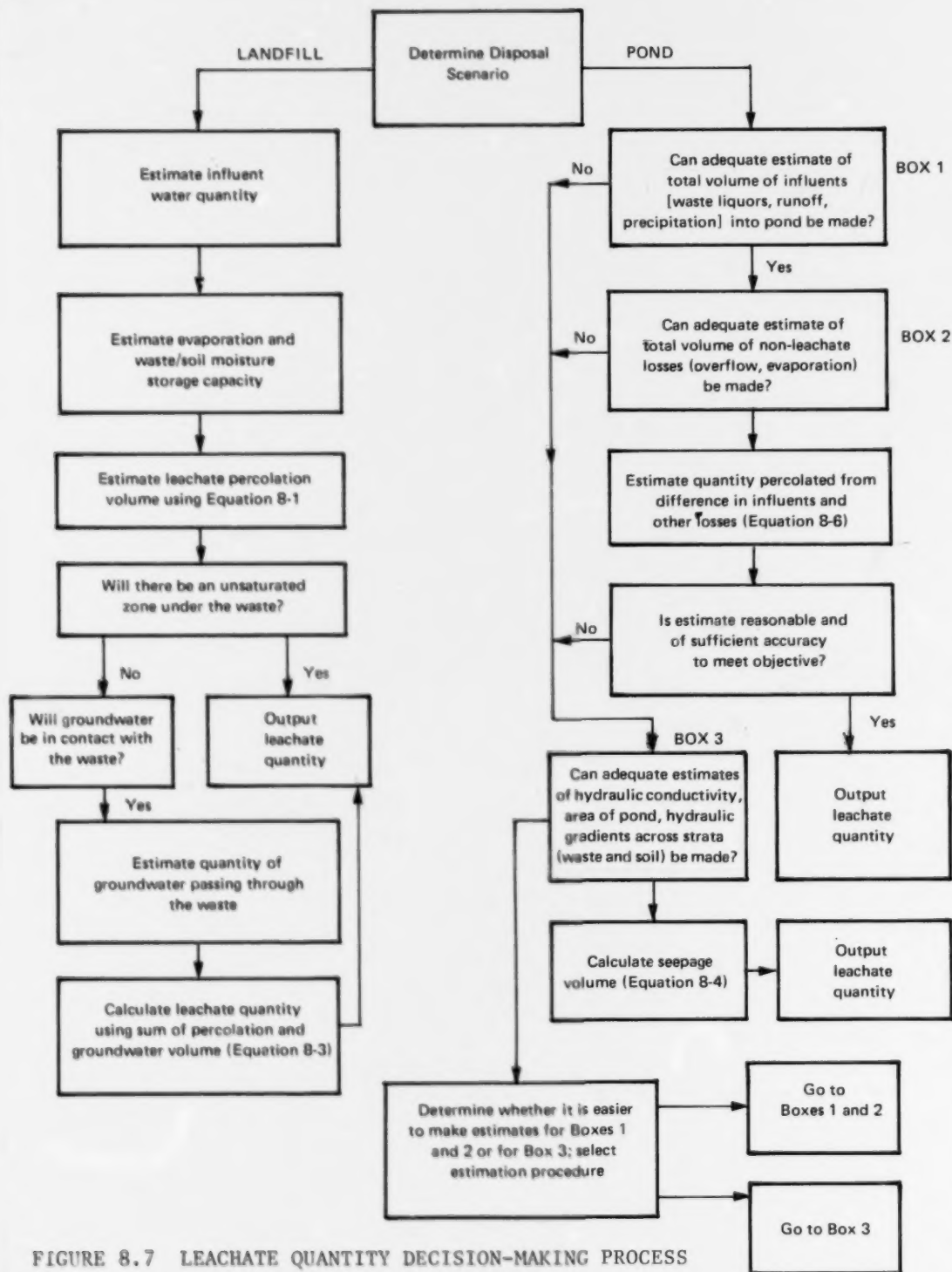


FIGURE 8.7 LEACHATE QUANTITY DECISION-MAKING PROCESS

liquors from different depths vary because of changes in the waste produced, liquid and solid contact time, as well as the number of "waste strata" the water has percolated through. Surface pond liquor composition may not reflect leachate composition in terms of readily leachable species that are not solubility limited. But in cases where the major species in the pond liquor have reached solubility limitations (i.e., some FGD ponds), their concentrations in the leachate can be quite similar to the pond liquor composition.

8.6.2.2.3 Dilution/Mixing of Interstitial Liquors -- For partially saturated wastes that contain residual process liquor (as in landfills), leachate generation, which occurs after influent water - rain or groundwater - locally saturates the waste, would cause the influent liquor to be admixed with the residual liquor. Thus, the quality of this mixture may provide the starting point for determining leachate composition. However, the original liquor may be diluted by mixing with several components (depending on the rain and groundwater chemical composition), and solubility limitations originally present in the residual process liquor may not prevail in the mixed liquor. Also, the ionic strength and other factors (such as buffering capacity, pH, complexing ion concentrations) of the resultant mixture may be modified to change leaching ability of that solution relative to the original liquor. Thus, depending on the extent of "dilution," the final leachate may be quite different from the original liquor. The extent of modification is related to many factors that affect the degree of leaching from the waste.

In some cases, waste saturation may not be necessary before liquor (leachate) is forced out of the waste. This could happen when all the pores where the entrapped air could escape from are filled with liquor.

8.6.2.2.4 Dissolution of Readily Available Species -- Readily soluble species in FGD wastes may include the inorganic major and minor components of FGD solids (e.g.,  $\text{Ca}^{2+}$ ,  $\text{SO}_4^{2-}$ ,  $\text{Cl}^-$ ,  $\text{Na}^+$ ,  $\text{K}^+$ ,  $\text{Mg}^{2+}$ ), major species with limited solubility that arrive to their saturated levels within relatively short periods of contact time (such as  $\text{CaSO}_3$ ), and trace components that are readily available to the liquid phase due to their preconcentrations on surfaces of the waste (as on ash) or their presence as readily soluble pure fine particles. In general, most solid phase species that are not encapsulated by other solid phases (see Section 8.6.2.2.5) may be considered readily available species. These can be more rapidly leached from the waste than those species limited by matrix dissolution.

Concentrations of these species can thus be limited either by their solubility in the particular liquid phase or by the total amount present in the solids. If the total amount present were the limiting factor, the ratio of liquid to solid in the disposal site would affect the final concentration. If solubility were limiting, the ratio will have no effect.

8.6.2.2.5 Dissolution of Matrix Species and Associated Trace Components -- Many factors can limit the rates at which trace components are leached from FGD and ash wastes. These include:

- Bulk matrix solubilization due to encapsulation (i.e.,  $\text{CaSO}_4$  and aluminosilicate matrix).
- Diffusion through hydrated layers of waste particles.

Leaching rates of encapsulated trace components may be increased by variables that raise the rate of bulk matrix solubilization. These variables include temperature and chemical composition of leaching fluid (pH, ionic strength, weak and strong complexing agents). The total quantity leached into solution will also depend on the contact time of the liquid and solid. Concentrations may be limited after a given period of time by adsorption from solution onto the solid and by solubility constraints.

Thus, to estimate the extent of leaching of trace components, their distribution in the waste (i.e., as surface concentrated, bulk matrix, encapsulated species, readily available) must be known. Laboratory tests may be required to provide this information.

8.6.2.2.6 Solubility Limited Species -- The major components of FGD waste ( $\text{CaSO}_4$  and  $\text{CaSO}_3$ ) result in levels of calcium, sulfate, and sulfite that are limited by their solubility product in the particular ionic strength and chemical composition of the liquor. Concentrations can be determined based on thermodynamic calculations and estimates of solubility. Solubility limitations are unlikely for other ash and FGD waste components, such as very soluble sodium and potassium salts. Some trends have been observed for certain trace elements (such as lower solubility at high pH due to formation of hydroxides), but it is difficult to predict routinely trace metal solubilities in various media.

8.6.2.2.7 Concentration Limitations Due to Availability -- Leachate concentrations of components that are readily and totally leached from wastes are determined by the quantity available from the solid and the liquid-to-solid ratio. Another contributing factor is the amount of "fresh" unleached waste that contacts this leachate as it percolates through the waste disposal site. Examples would be ash particles where all of the As and sulfate are present on the surface and, in some cases, Na and K salts.

8.6.2.2.8 Speciation of Elements in Leachates -- Speciation in leachates is discussed here in two parts. The first is oxidation states of particular elements; the second is formation of complexes in solution. Speciation is important both in terms of environmental effects (for instance, certain oxidation states of As are more toxic than others) and further reactions in the environment (such as attenuation).

The oxidation state of many elements in leachates under normal conditions is defined by the presence of only one stable form of the element (i.e., Na,  $\text{Ca}^{+2}$ ,  $\text{Cl}^-$ ,  $\text{Sr}^{+2}$ ). For other elements, (notably transition elements and As), various oxidation states are possible. Since experimental data on speciation of elements are lacking, most of this discussion is based on the expected chemistry of particular elements under conditions of solution redox potential. Direct measurements under actual field conditions provide the best estimates,

but such tests are difficult. Most measurements to date on trace metal concentrations in leachates provide total element concentrations with no attempt at speciation.

Given a leachate chemical composition and oxidation/reduction potential, the most stable thermodynamically oxidation form of an element can be defined. However, the kinetics of the rate required to achieve that thermodynamically stable state may not be known if the species is not initially present in that state. Thus, both  $\text{As}^{+3}$  and  $\text{As}^{+5}$  have been determined under reducing and oxidizing conditions in ash pond leachates, while  $\text{Cr}^{+6}$  has been observed in extracts of some FGD wastes. The oxidation states of readily oxidizable ions ( $\text{Fe}^{+2}$ ) under weakly complexing environments will tend to the higher value ( $\text{Fe}^{+3}$ ) under aerobic (oxidizing) conditions. Similarly under anaerobic conditions, bacterial action may cause the reduced oxidation state. Interelement redox reactions can also occur.

Complex formation occurs to a greater extent in leachates of higher ionic strength than in dilute solutions. In any case, most complexes formed in these leachates are expected to be "kinetically labile." That is, rates of conversion of the complex to another form are extremely rapid when the chemical composition of the leachate media is changed. For example,  $\text{Ca}^{+2}$  may form  $[\text{CaCl}^+]$  and  $[\text{CaSO}_4]$  complexes in solution, but when the leachate is diluted with water under the site, these complexes will rapidly form a new equilibrium concentration. This is important, since subsequent reactions will be determined by speciation in the solution undergoing reaction and not by the original leachate.

Experimental determination of these types of complexes in solution has not been performed for FGC waste leachates. Computer programs are available that use thermodynamic complexation constants to predict the relative concentration of a particular species, given the total element concentration and complexing species present in solution.

8.6.2.2.9 Leachate Chemical Quality Decision Tree -- The purpose of the leachate chemical quality decision tree (Figure 8.8) is to estimate the concentrations (and speciation, if needed) of particular elements in the leachate that may exist in the disposal site.

Mobility of waste-related species occurs by migration of aqueous phases. Thus, composition of the aqueous phase in the waste itself as well as those of the aqueous phases generated by contact with leaching media are important in determining environmental effects. The initial step in determining leachate quality would be to correlate the specific disposal scenario with available data for similar situations. This may provide enough information. If not, more detailed data may be obtained, as shown in Figure 8.8.

If the waste has an aqueous phase (as in a pond), the primary leachate composition will be related to that of the interstitial liquor. Since the chemical properties of the waste will vary within a given disposal site, different spatial distributions and changes with depth can be expected. In addition, site-specific conditions, such as the redox environment may cause

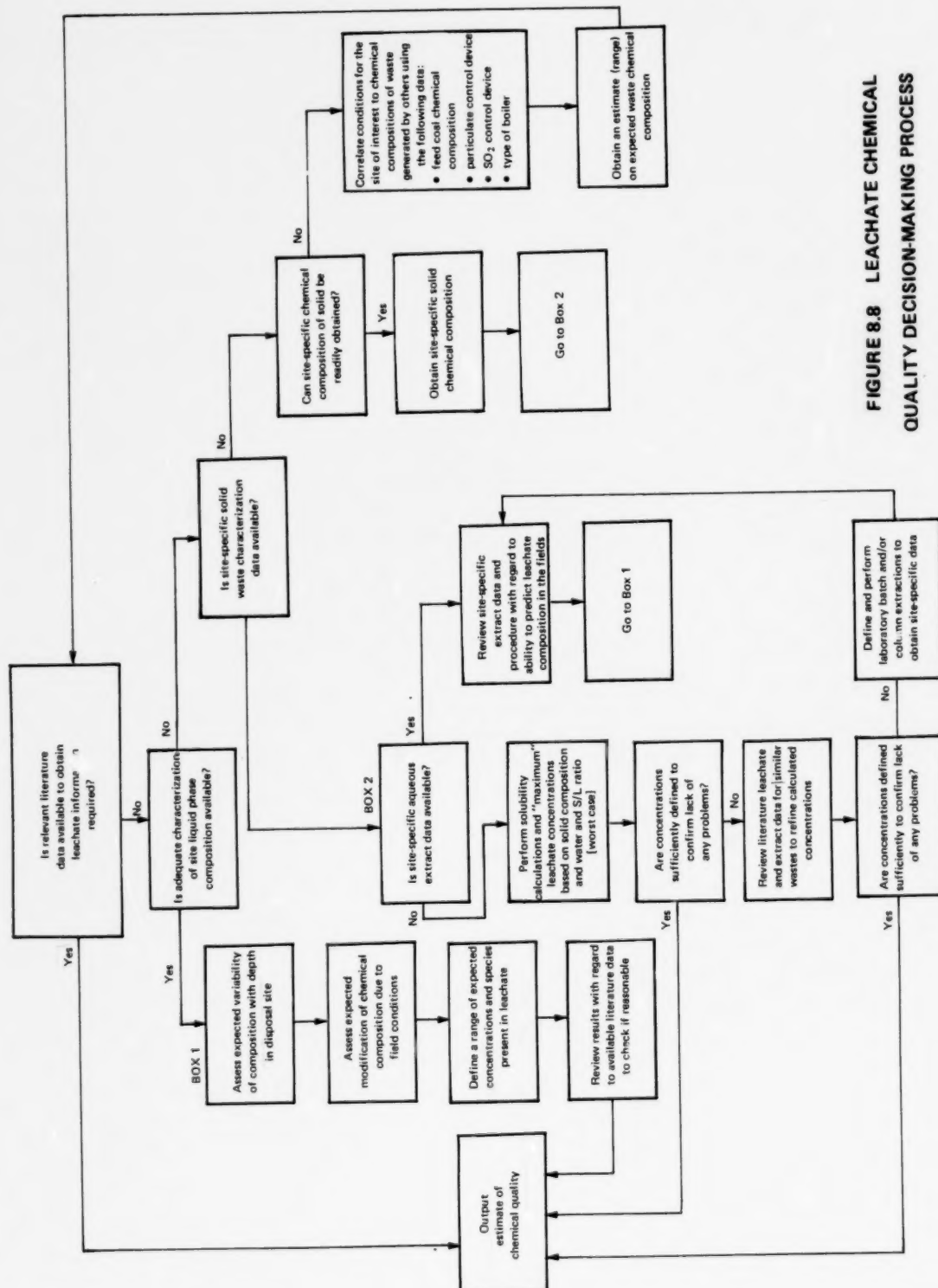


**FIGURE 8.8 LEACHATE CHEMICAL QUALITY DECISION-MAKING PROCESS**



variations of both concentration and speciation between surface samples and interstitial waste liquors. Also, because liners retard water movement, the liquid-to-solid contact time can be increased, resulting in more dissolution and hence higher concentrations of trace species. Thus, although an initial estimate of the liquid phase composition may be available, the effects of site-specific conditions must be considered.

If no site-specific data on waste liquor are available, the next best source is the composition of the waste solid phase. Given data for solid composition but no extract data for leachate concentrations, a three-step process can be applied:

- Leachate "worst case" levels can be initially estimated by using available thermodynamic data to calculate maximum solubility of waste species in the leaching media. If species are unlikely to be limited by solubility, the maximum level can be defined by assuming that all of the element is leached for each waste layer the water contacts as it percolates through the site. This step may be sufficient for some questions.
- If a better estimate is required, literature data on comparable extracts of the particular waste can be used to estimate the rate of leaching. This step may be necessary when the experimental data suggest that only a small fraction of the total present is available for leaching in a reasonable period of time, or observed leachate/extracts are very different from calculated levels.
- If further refinement is desired, site-specific laboratory and field tests may be needed.

If sufficient extract data for the waste are available, this three-step process should be undertaken before the data are used. When no adequate solids data are available, two options are open. One approach is to determine composition by conducting experiments. The other is to estimate the composition of the waste solids by comparison with less specific literature data for similar disposal conditions.

#### 8.6.3 Leachate Admixing

Leachate admixing refers to the physical and chemical processes whereby leachate concentrations are changed by admixing with receiving waters. This discussion focuses on physical processes that may be readily evaluated in decision-making by examining chemically conservative leachate species. If the receiving water shows negligible concentrations of the constituents in question, the admixing process may be one of simple dilution. In several environmental settings, however, the preexisting levels for constituents generally associated with utility waste leachate may be elevated in receiving waters. This means that the admixing process will only have a negligible dilution effect. In general, where receiving waters exhibit comparable, or greater, levels of species of interest than the leachate, little or no adverse impacts would result from the disposal activity. At other sites where



leachate is admixed with relatively larger volumes of receiving water, examining leachate generation, movement, and admixing processes may by itself indicate negligible adverse effects on water quality. In such cases, there would be no need to study the chemical attenuation processes before admixing.

Evaluating the effects of admixing generally requires an understanding of leachate flow rates and quality, receiving water flow rates and quality, and the nature and rates of dispersion and chemical fate processes that control the admixing process. It is be important to define the location at which the admixed concentration is desired (i.e., drinking water well, critical habitat location directly proximate to the waste) as well as a representative volume of the receiving water body (i.e., aquifer, stream) over which the concentration of species of interest may be averaged. These considerations are discussed below.

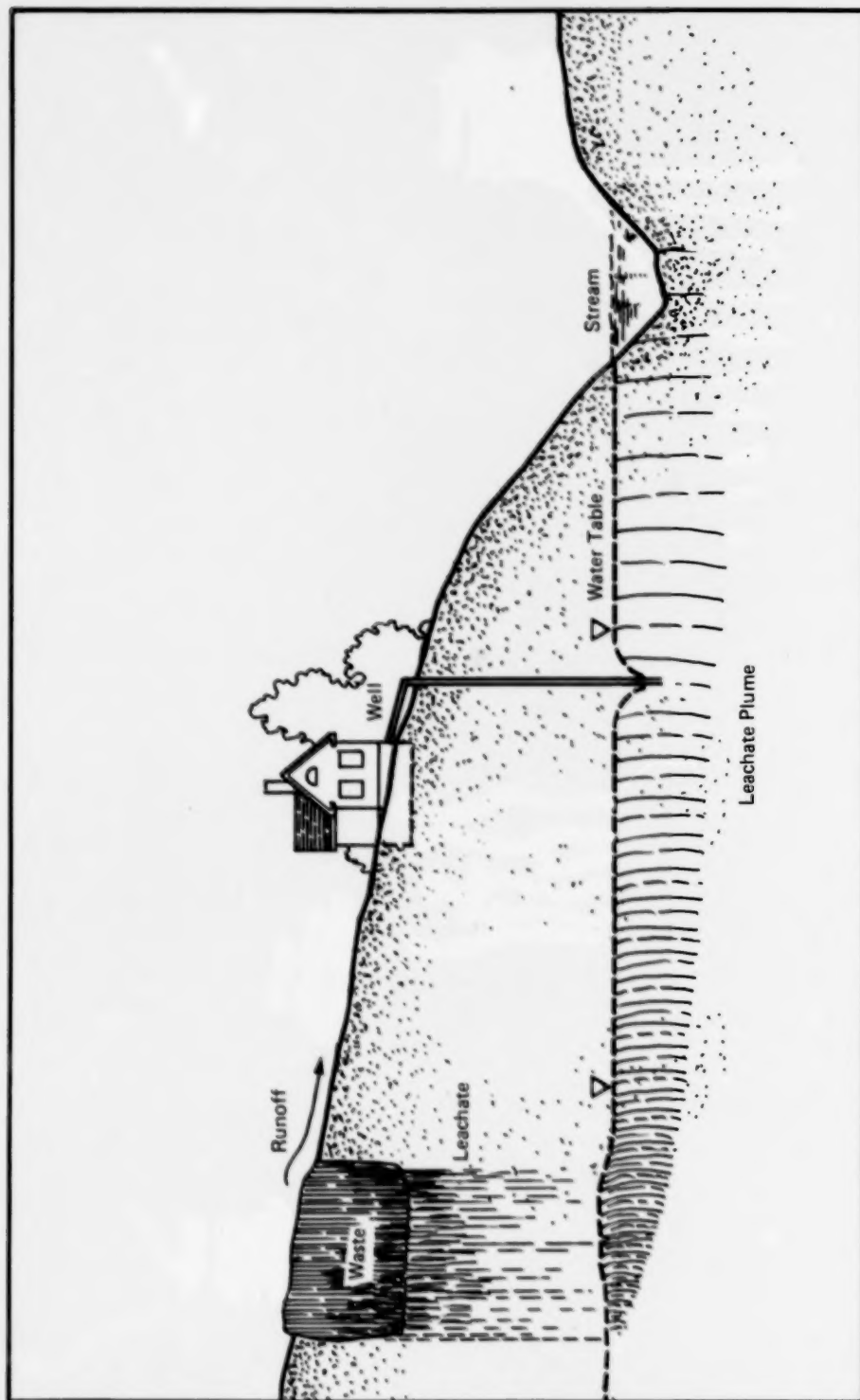
#### 8.6.3.1 Admixing in Groundwaters--

Consider the simplified disposal site environment depicted in Figure 8.9. Leachate percolates down to the water table, laterally toward a stream discharge point. (There may be no unsaturated zone under some sites. This will affect leachate movement and admixing, and is discussed separately.) For the situation shown in Figure 8.9, vertical movement in the aquifer is assumed negligible relative to horizontal movement. This condition tends to prevail if leachate movement rates are small compared with lateral underflow rates of the aquifer.

First, one must consider whether vertical or lateral gradients of leachate contaminants in the aquifer are significant. Dispersion, the tendency of solutes to spread out from the predicted path according to hydraulics of flow, can usually be ignored under the following circumstances:

- The disposal site is immediately adjacent to a groundwater discharge area - a stream or lake bed.
- A production well that draws water from the full saturated thickness of the aquifer is or will be downgradient of the disposal site.

In both cases, the contaminant will be rapidly admixed at the discharge point. At a natural discharge, such as a stream bed, a shallow aquifer almost invariably discharges all of its flow with relatively rapid admixing in the surface water body. A production well also mixes the groundwater during the drawdown. At maximum sustainable yield, such wells draw from the full aquifer thickness. A well that is screened or otherwise open through the full saturated thickness will also artificially mix the groundwater in the open portion of the well itself, regardless of pumping rate. In considering the effect on production wells that may be developed in the future, one must account for their zone of influence in determining whether the well will be "downgradient" of the site after development. In each case, the admixed concentration can be estimated by assuming that leachate mixes through the full depth of the aquifer.



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FIGURE 8.9 GENERAL WASTE DISPOSAL SETTING

Dispersion effects need not be explicitly accounted for if the contaminant will be well mixed throughout the aquifer thickness at the location where concentration estimates are desired and if that location is roughly in the "middle" of the contaminant plume (see Figure 8.10). But dispersion phenomena must be implicit in this determination. The "width" of the fill,  $b$ , is defined directly upgradient from the site. The plume width,  $W$ , will roughly increase along downgradient flow lines as:

$$W = b + 6\left(\sqrt{2 \alpha_T X}\right) \quad (9)$$

where

$X$  is the distance downgradient (along a flow line) and  $\alpha_T$  is the transverse dispersivity.

The vertical spread of the plume,  $T$ , is approximated by:

$$T = 3\left(\sqrt{2 \alpha_T X}\right) \quad (10)$$

Measured values of  $\alpha_T$  have been reviewed by Anderson and may range from approximately 0.01 to 10 m (74). Sykes et al., in a recent study of disposal site scale contaminant migration phenomena(75), recommend values of 0 to 0.15 m, suggesting that the larger dispersivities cited by Anderson may only be appropriate for plume migration at larger spatial scales. Values of 0.01 to 1 m seem appropriate based on the available information.

If  $T$  is greater than aquifer thickness, uniform vertical mixing can be assumed. For example, at the new Elrama landfill studied under this project (see Section 5.0), leachate may seep vertically to an alluvial aquifer that has a saturated thickness of about 8 m (25 ft). Looking directly upgradient, the fill intercepts about 800 m (2,500) feet of aquifer width. The alluvial aquifer discharges to the Youghiogheny River at Elrama roughly 250 m (800 feet) downgradient of the fill. The width of the potentially contaminated portion of the aquifer at the discharge area is:

$$W = 2500 + 6\left(\sqrt{2(\alpha_T) 100}\right) \quad (11)$$

where  $W$  = width in feet.

If  $\alpha_T = .03$  to  $3.0$  ft,  
then  $W = 2540$  to  $2860$  ft

Thus, lateral dispersion causes a small amount of plume spreading, which in this case can be ignored. The vertical spread of the plume is given by:

$$T = 3\left(\sqrt{2(\alpha_T) X}\right) \quad (12)$$

In a hypothetical situation where a small well were to be located between the fill and the river (30 m or 100 ft from the river),  $X = 152$  m (500 ft) and  $T = 5$  to  $50$  m (16-160 ft). Compared with the saturated 8 m (25 ft), it is likely that the plume will be well mixed vertically through the full aquifer thickness.

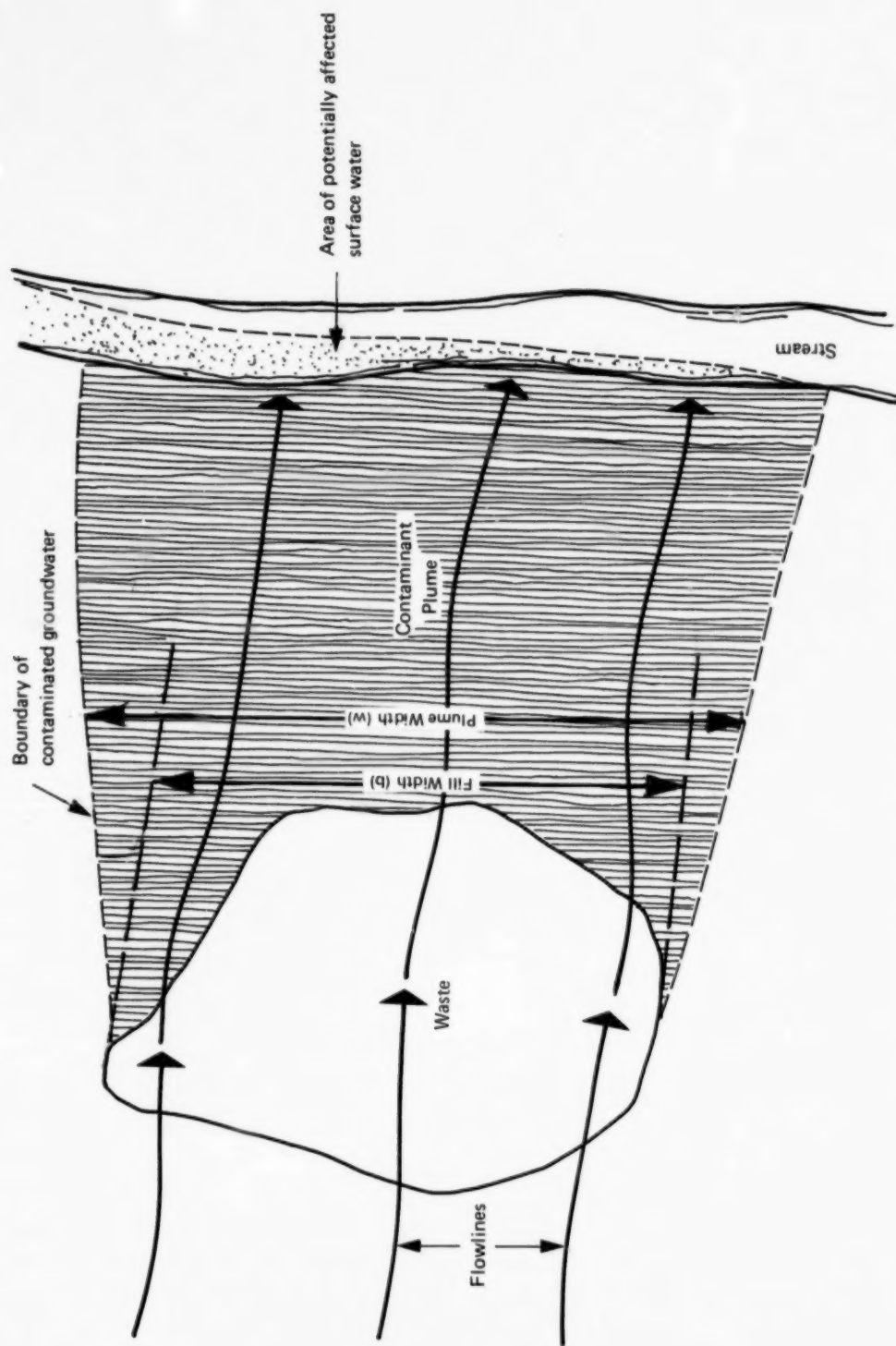


FIGURE 8.10 PLAN VIEW OF GENERALIZED CONTAMINANT PLUME

Once the depth and width of the plume at a sampling location are estimated, the mixing ratios may be determined. The first step is to define the leachate generation rate,  $Q_1$  ( $m^3/yr$ ) (see Section 8.6.2), and the flow in the affected part of an underflowing aquifer,  $Q_a$  ( $m^3/yr$ ). The flow  $Q_a$  is through a cross section of width  $W$  and thickness  $T$ . To estimate  $Q_a$ , multiply the Darcy velocity by  $W$  and  $T$ .

Within approximately a thousand feet of a typical disposal site, lateral dispersion can be neglected. Also, receptors have been identified, such as lakes and streams, large production wells, and wells screened (or otherwise open) over the full saturated thickness, for which limited vertical dispersion need not be considered and  $T$  equals the thickness of the saturated zone. In other instances,  $T$  can be estimated as:

$$T = 3 \left( \sqrt{2 \alpha_T X} \right) \quad (13)$$

where

$T$  cannot exceed  $h$  (the saturated aquifer thickness)

Then, the admixed concentration of any constituent,  $C$  is given by:

$$C = \frac{C_1 Q_1 + C_a Q_a}{Q_1 + Q_a} \quad (14)$$

where

$C_1$  is concentration in leachate

$C_a$  is concentration in background groundwater

$Q_1$  and  $Q_a$  are quantities of leachate and groundwater, respectively

The situation is more complex when leachate is a substantial contributor to downgradient flow in the aquifer. This often occurs with unlined disposal ponds, where the downward leachate flow forces contaminated water to greater depth than would occur from dispersion alone. This problem is addressed carefully by McWhorter & Nelson (76) and will not be reviewed here. Given an estimate of vertical and lateral linear flow velocities ( $V_z$  and  $V_l$ , respectively), a crude estimate of the depth of leachate mixing ( $T$ ) may be made:

$$T = \left( \frac{V_z}{V_l} \right) X \quad (15)$$

where

$X$  is distance taken along the flow path

As an example calculation from the studies at the Allen plant (see Section 5.0),  $V$  is roughly  $10^{-6}$  cm/sec,  $V_l$  is roughly  $10^{-5}$  cm/sec and  $T \approx 0.1 X$ . Since the saturated thickness of the aquifer is roughly 18 m (60 ft), the leachate will be well mixed vertically through the full aquifer thickness within 182 m (600 ft) of the pond center (well within the bounds of the pond



itself). In this case, limited vertical dispersion need not be considered. Contaminants will be well mixed through the full aquifer thickness.

#### 8.6.3.2 Admixing in Surface Waters--

Contaminated groundwater usually mixes so rapidly with surface water bodies that no discernible "hot spots" of contamination are expected. Findings at the Powerton site (Section 5.2.5) demonstrate that significantly contaminated groundwater discharging to a very small stream causes only minor observable water quality changes in the stream. Available data, however, cannot discount the possibility of surface water effects from a large, unlined pond discharging to a small stream or to a quiescent body of water such as a small lake or meandering river.

Leachate admixing in streams may be estimated as follows. Vertical mixing is assumed essentially instantaneous; lateral mixing is not. A useful rule-of-thumb (derivable from stream dispersion coefficients) is that if a pollutant is introduced at the shoreline, it will be well mixed across the full width of the stream at a distance  $X = 10 W_s^2 / d_s$  downstream from the source, where  $W$  and  $d$  are the stream width and average depth, respectively. Downstream of this point, contaminant concentrations in the stream may be estimated as:

$$C = C_{up} + \frac{L}{Q_s} \quad (16)$$

where

$C_{up}$  is the upstream or background concentration (mg/l),

$L^{up}$  is the contaminant loading rate (mg/sec),

$Q_s$  is the stream flow (l/sec).

Upstream of this point, where more acute impacts may be expected, a worst case estimate may be made by assuming the contaminated groundwater discharge is a point source, well mixed vertically. The simplified approximation is:

$$C = C_{up} + \frac{L}{\sqrt{\frac{d}{10} \frac{X}{W_s^2}} Q_s} \quad (17)$$

This equation yields unnaturally high estimates for very small values of  $X$ , because of the unrealistic assumption that the contaminated groundwater discharge is an ideal point source. This may be avoided by ignoring any estimated concentrations that exceed the concentration in the contaminated groundwater. The concentrations estimated by this equation do not apply to the whole stream, but only to a region near the discharge point (shore or stream bed), with approximate lateral dimension given by:

$$\sigma_y = X \left( \sqrt{\frac{d}{10}} \right) \quad (18)$$



#### 8.6.3.3 Summary--

The methods suggested here are appropriate only under relatively simple and well characterized geohydrologic conditions. Even then they are only reliable to within about an order of magnitude. This level of accuracy, however, may frequently be adequate for an appropriate waste management decision. A really accurate understanding of groundwater flow patterns can only be based on careful field surveys. Simple calculations or sophisticated computerized models cannot compensate for basic deficiencies in the geohydrologic description of a site. Indeed, excessive reliance on models can obscure an often inadequate understanding of site specific geohydrology. However, as mentioned in Section 8.6.3.4, computer modeling may be appropriate in some situations. Several models may be used. Available sources include EPRI, USGS, the Groundwater Model Clearing House (operated by Holcomb Research Institute of Butler University and National Water Well Association), and References 77 and 78.

#### 8.6.3.4 Leachate Admixing Decision-Making Process--

Figure 8.11 shows a methodology for evaluating the effects of leachate admixing with receiving waters. The receiving waters may be surficial (streams, rivers, lakes, estuaries) or subsurface aquifers. In the most general and typical case of disposal near a surface water body, one must consider admixing of leachate with underflowing groundwater, followed by admixing of this contaminated groundwater with the surface water body as the groundwater discharges to it.

The first step is to compare leachate quality with receiving water quality for all constituents of interest. The list of constituents will generally include major ions and heavy metals. Leachate quality may be comparable to or better than receiving water quality for many important constituents in several settings: (a) highly mineralized locations in the arid west, (b) highly acidified locations in the interior, and (c) tidally-influenced coastal settings. Preexisting site contamination is another factor when assessing surface mine disposal operations.

The next step is to review the site water balance and other hydrologic and geohydrologic data to determine the range of flow rates expected in the receiving waters over the life of the facility. Seasonal variability in flows should be considered, as well as the substantial uncertainties inherent in estimating groundwater flow rates. The uncertainty in groundwater flow rates, as estimated either from available data or limited site field survey, may be quite high (as much as an order of magnitude). Estimates are likely to range over a factor of 0.2 to 5.0 times the value. These wide ranges should be accounted for by conducting alternative admixing analyses based on the extremes of the best estimate, if no independent estimate of uncertainty is available.

Seasonal variations in receiving water flows are likely to have a substantial influence on the admixing process in surface waters. Admixing at low flows can be calculated based on the seven-day, ten-year low flows that are available for most major streams and rivers from local, state, or federal water pollution control agencies for the Water Resources Division of the

C = constituents  
RW = receiving water

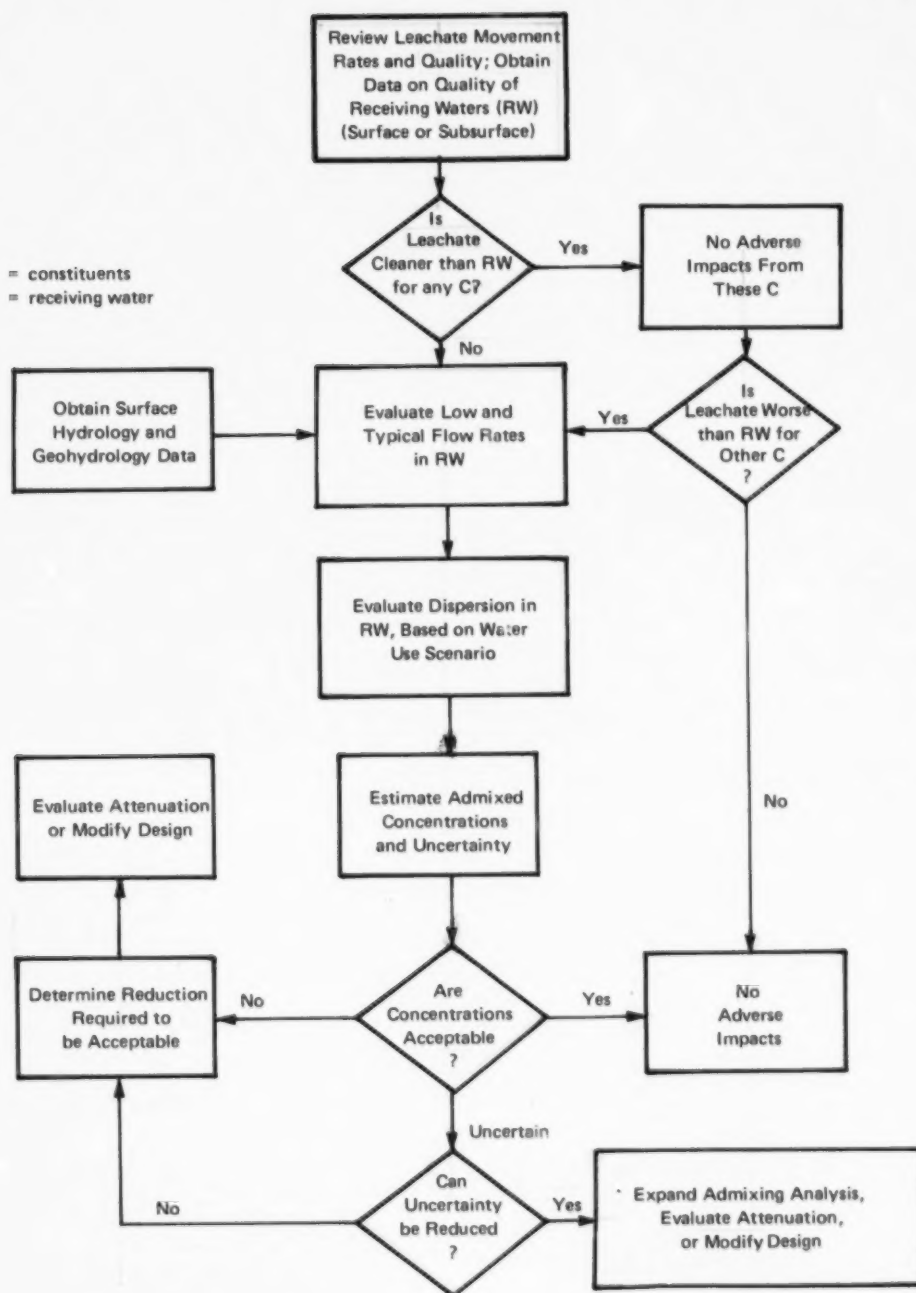


FIGURE 8.11 LEACHATE ADMIXING DECISION-MAKING PROCESS

District Office of the U.S. Geological Survey. If the seven-day, ten-year low flow has not been defined, it can be estimated from a suitable hydrologic time series (79). When suitable data are not available for a direct determination of a seven-day, ten-year low flow for the water body, a variety of indirect methods may be used to define a representative low flow. Low flow can be estimated based on data on the low flow of a nearby stream whose drainage area most nearly corresponds to that of the subject stream in drainage area, climate, and physiography. A reasonable estimate would be that the ratio of the low flows in the two streams equals the ratio of the drainage area. Similarly, if the mean flow of the stream is unknown, the mean flow will be roughly proportional to drainage area, based on other nearby streams whose watersheds are similar in climate and topography.

The effect of seasonal variation in groundwater flow may be accounted for, assuming sufficient data are available. Shallow aquifers affected by surface disposal may show substantial seasonal fluctuations in flow and head. The admixing process, however, may be fairly insensitive to seasonal fluctuations for the following reasons:

- Groundwater admixing is a relatively slow process that occurs over the total travel time from source to receptor; this movement often takes several years and has an averaging impact on seasonal fluctuations.
- At some disposal sites, particularly landfills, fluctuations in the amount and movement of leachate may match those of the groundwater. The resulting concentrations may be relatively insensitive to seasonal variations.

Seasonal fluctuations in groundwater admixing need only be considered when the travel time to significant downgradient receptors is less than one year. In interpreting field data to estimate typical groundwater flow rates, however, it is important to evaluate the degree to which a particular observation period is representative of typical flow rates. If data are available from a limited observation period, geohydrologic/climatic interpretation of the representativeness of the observation period is recommended.

Data characterizing leachate flow rates and quality, receiving water body flow rates and quality should be compiled and compared with leachate and receiving water quality to screen constituents for which leachate quality is better than receiving waters. The final steps of the methodology lead to an estimate of receiving water quality at locations of sensitive water use, assuming that chemical attenuation has not occurred. If the estimated concentrations are within acceptable limits, the chemical attenuation processes would not have to be evaluated, as attenuation would likely reduce concentrations further. With some constituents, however, such as acidic leachates, interaction of the soil with leachate can lead to elevated levels and would thus require attention.

The final steps are the most difficult and controversial in evaluating admixing. Problems arise when data bases and field and theoretical methods

designed for water supply purposes are applied to pollution migration problems at disposal site-scale. For example, small-scale inhomogeneities in properties of the transport medium can control actual pollution migration pathways and apparent dispersion, thus determining whether contaminant will affect a specific well location. Such inhomogeneities are generally not characterized in detail during standard site evaluation field work, nor are models in general use designed to account for their effects on pollution migration pathways. For these and other reasons, it is not possible to predict precisely the contaminant concentration distribution that will result from a specific waste disposal configuration.

Detailed computerized modeling of groundwater contaminant migration at utility disposal sites is not well enough developed to be recommended as standard practice. Use of such models would not substantially reduce uncertainties in predicted groundwater quality, although some exceptions may arise:

- If a calibrated groundwater flow model is already available for the site. (Such a model may have been developed for utility water supply or other purposes). Computerized prediction of contaminant migration patterns may be achieved at small additional cost.
- If noncomputer-based evaluation procedures leave too much uncertainty regarding the seriousness of environmental impacts, and this uncertainty may be reduced computerized models. This case can only be evaluated when the uncertainty of the evaluation procedures is carefully estimated, and the bounds of uncertainty encompass a critical environmental quality objective (such as a drinking water standard).

Computer modeling could be appropriate where the geohydrologic setting is complex or subject to significant future stresses, and where that complexity is well characterized by field data. But models cannot be used to overcome data deficiencies. For example, a potential disposal site is underlain by a sand and gravel aquifer which is separated into two weakly interacting strata by a clay bed. The bed is present in most borings but absent in a few. A municipal production well serving domestic users draws from the lower stratum. Noncomputer-based evaluation procedures are not well suited to this complex configuration, and it is quite uncertain whether leachate constituents will appear in the well water at hazardous concentrations. Computerized modeling may be necessary to evaluate this situation. Other examples include those where future stresses on the hydrogeologic system (a planned production well or substantial additional recharge from a disposal pond) will substantially change groundwater flow processes.

Dispersion phenomena in the receiving waters should next be considered. Specific methods have been presented in the previous section. This step should consider the volume of the receiving water that is actually significant in the context of a specific water use. For example, it is not appropriate to estimate the concentration in a portion of an aquifer one meter cubed in



volume if a municipal well draws from the full aquifer thickness with a cone of influence of one hundred meters cubed in volume.

Once the extent of dispersion is estimated, concentrations may be crudely ascertained by assuming that concentrations are relatively uniform within an affected portion of the receiving waters. Uncertainty in the flow-related parameters must be accounted for, as well as any ambiguities in the water quality of leachate and receiving waters.

The resultant concentrations must be compared with environmental quality objectives. Depending on the level of uncertainty associated with these results, further field investigations or model applications may be necessary. If these measures will not reduce the uncertainty to an acceptable level, the only recourse is to see whether chemical attenuation would lead to a concentration reduction or whether the disposal plan should be modified.

#### 8.6.4 Mechanism of Attenuation by Soils

##### 8.6.4.1 Overview--

As leachate migrates out of the boundary of the waste disposal site, it contacts surrounding soil and moisture present in that soil (whether groundwater or unsaturated soil moisture). During this contact, chemical and physical reactions can occur between leachate chemical species and the soil/water mixture. If these reactions lower concentrations of chemicals in the mobile liquid phase, they are considered capable of attenuating species from the leachate. This section summarizes mechanisms governing leachate attenuation by soil/water mixtures and the properties of each component that may affect the extent of attenuation.

##### 8.6.4.2 General Mechanisms--

The general mechanisms that may reduce soluble component concentrations in leachate include:

- dilution of leachate by existing liquid or moisture in the soil;
- precipitation of species in the leachate by modification of the leachate chemical composition after mixing with solid soil components and with liquid or moisture present in the soil;
- adsorption of chemical species from the leachate/water mixture onto surfaces of soil particles; and
- formation of insoluble complexes between species in the leachate and complexing agents in the soil.

Dilution of the leachate by existing groundwater in the soil (or soil moisture) can cause reductions concentrations observed in the leachate. Where unsaturated soils occur under the waste disposal area, this mechanism may not be significant until the leachate reaches the groundwater level. But if the soils are saturated (i.e., groundwater is directly under the waste or in the waste), it may play a larger role (see Section 8.6.3).

Species can precipitate from the leachate due to changes in leachate pH which occur as it mixes with underlying soil and/or water. Many trace metals are more soluble at lower pH's. An acidic leachate may contain elevated levels of trace elements which will precipitate if the leachate mixes with alkaline water in the soil or with alkaline soil itself. Also, the mixing can cause levels of particular components (i.e.,  $\text{SO}_4$ ) to increase in the mixture. This, in turn, can lead to precipitation of soluble cations via the "common ion" effect. A combination of effects due to changes in both pH and chemical composition may cause lower solubilities of various solid phases. Additional reactions (i.e., redox) between soil solids/liquids and leachates can modify speciation to produce less soluble compounds.

Species from the resultant leachate solutions can be adsorbed onto soil particles due to ion exchange-type interaction (opposite charges) or simply Vander Waals-type forces. This interaction depends highly on the surface composition of the soil and surface area of the particles. In general, hydrous metal ion hydroxides (Mn, Fe and Al) play a large role in adsorbing trace elements onto the soil. The greater the surface area (greater proportion of small clay type particles), the greater the expected adsorption. Leachate composition is significant in this equilibrium, since the species present in the solution will largely determine the extent of adsorption. All chemical species present in solution which affect speciation of the particular element and those that influence the charge on the soil particles are important.

Formation of insoluble metal complexes between humic acids in the soil and particular trace elements has also been proposed as a mechanism for attenuation. As in adsorption, the composition of the leachate in terms of speciation and modification of charges on soil particles can affect the interaction with complexing agents in the soil. The presence of "free" complexing sites is influenced by leachate pH and the presence of other soluble constituents. The tendency for a trace component to be complexed also depends on its speciation.

Additional information on these mechanisms and a summary of available data are provided in References 80 and 81 and in the results of the soil attenuation experiments conducted in this study (see Appendix F).

#### 8.6.4.3 Properties of Soils Affecting Attenuation Capability--

8.6.4.3.1 Overview -- This section summarizes the properties of soil that affect their attenuative capacity. The important physical and chemical properties are:

##### Physical Properties

- particle size distribution (% clay, % silt, % sand)
- surface area (measured experimentally or calculated based on particle size distribution)
- moisture content



### Chemical Properties

- total organic carbon
- pH of soil/leachate mix (available alkalinity)
- chemical composition of moisture present
- chemical composition of major and trace components

Sorption of a given chemical species varies greatly from one soil to another. Similarly, sorption of different species on one particular soil fluctuates widely. However, greater surface areas (smaller particle sizes) generally lead to greater attenuation. Dilution (high moisture content in the soil) provides an initial "attenuation." Greater organic carbon content, higher pH (leachate/soil mixture pH about of 7 to 9), and higher concentrations of hydrous oxides Mn and Fe increase soil capacities for many trace components.

Release of species from the soil into the leachate when the two come into contact is also observed, especially where the leachate/soil mixture is highly acidic (pH 2 to 4) and also where the leachate contains elevated concentrations of species which can displace substances from the soil. Field parameters that affect leachate chemical quality (composition, permeability, moisture content) are also important in determining the extent of attenuation. Adsorption phenomena usually occur in a short time frame (less than 1 day) under good mixing conditions. In field situations where little mixing occurs, adsorption rates may be limited by the slower diffusion of species in aqueous phases. Temperature may also determine the extent of attenuation. Differences between the temperature in the field and the temperature at which laboratory experiments have been performed can lead to erroneous assumptions regarding behavior in the field.

One area that represents a large data gap concerns changes in the chemical composition of the soil/leachate mix. More information is needed on how these changes influence speciation.

#### 8.6.4.4 Properties of Leachate Affecting Attenuation Capacity--

Chemical properties of leachates that can be important in attenuation include:

- concentration (activity) of species of interest in solution,
- speciation of element of interest, and
- concentrations (activities) of other components in solution.

The most obvious leachate property determining soil capacity for a particular species is its concentration in the solution. The capacity of the soil rises as the concentration in solution increases, until the soil is saturated with regard to its capacity. Thus, lower concentrations in solution

would lead to lower capacities. In many cases, plots of log (concentration in the soil -  $\mu\text{g/gm}$ ) versus log (concentration in solutions -  $\mu\text{g/ml}$ ) are linear according to the type of relationships derived by Freundlich (14).

Speciation of an element is important in adsorption and precipitation reactions. Speciation includes the oxidation state of the element and its state of complexation. Highly complexing species in solution may associate with the element to form stable complexes with different affinities for the soil than the "uncomplexed" species. Variation in pH can produce stable hydroxy-type complexes (i.e.,  $\text{H}_2\text{AsO}_4^-$ ), which will be attenuated differently from  $\text{AsO}_4^{3-}$ . A large concentration of  $\text{Cl}^-$  or  $\text{SO}_4^{2-}$  can produce ion pairs or complexes in solution ( $\text{CaCl}^+$ ,  $\text{CaSO}_4^0$ ) which may reduce the extent of attenuation. Little experimental data are available on the speciation of substances in actual samples, although many predictive calculations have been made. Current data for speciation of elements in coal ash and FGD waste leachate are limited to some determinations of oxidation states rather than presence of complexes. The presence of 10 to 20 ppb of a strong organic complexing agent in the leachate can complex all of a particular element present at low concentrations (i.e., Cu at 10 to 20 ppb) and prevent any attenuation at that level.

Concentrations of other species in the leachate soil mixture or solution can influence attenuation by: causing changes in the speciation of the element of interest; providing opportunity for precipitation of species with various other ions; and competing for limited adsorption on soils. Other species, as well as pH, can also cause changes in the soil surface charges, thereby affecting adsorption capacity. In some cases, species can be released from the soil.

Most studies of how a given element affects adsorption capacities of various soils have been conducted with relatively "pure" synthetic solutions. In this study, some data has been obtained for actual leachates from FGC disposal sites. Speciation and affects of other species in solution on attenuation have received little attention and represent a significant data gap.

Table 8.4 shows the relative mobility of some trace elements (existing as cations and anions in solution) for a variety of soils.

#### 8.6.4.5 Soil Attenuation Decision-Making Process--

The decision-making process (Figure 8.12) may be undertaken to determine the extent that soil attenuation of leachate species occurs. The path for the most site-specific information assumes that chemical data on interstitial soil pore water directly under the site are available. These data are then adjusted (for mixing with liquids present in the soil or groundwater) and compared to the concentrations in the leachate to determine if the soil has caused changes in concentrations.

If adequate information on the composition of soil water under the site is not available, the composition of the soil solid phase (physical properties such as percent clay, silt, and sand and chemical properties such as total

TABLE 8.4  
RELATIVE MOBILITY OF ANIONIC AND CATIONIC METALS IN VARIOUS SOILS

| <u>Soil Type</u>   | <u>Low</u>                       | <u>Moderate</u> | <u>High</u>         |
|--------------------|----------------------------------|-----------------|---------------------|
| Clay               | Cu,Pb,Be,Zn,Cd,Ni,<br>Se,V,As,Cr |                 | Hg                  |
| Silty Clay         | Cu,Pb,Be,Zn,Cd,<br>Ni,Se         |                 | V,As,Cr             |
| Clay Loam          | Cu,Pb,Be,Zn,Cd,Ni                | Se,V            | As,Cr               |
| Silty Loam         | Cu,Pb                            | Be,Zn,Cd,Ni,Hg  |                     |
| Silty Clay<br>Loam | Cu,Pb                            | Be              | Zn,Cd,Ni,Hg         |
| Silt               | Cu,Pb                            | Be,Se           | Zn,Cd,Ni,Hg,V,As,Cr |
| Silty Loam         | Cu,Pb                            |                 | Be,Se,V,As,Cr       |

Note: Se,V,As,Cr are considered to exist as anionic complexes in the solutions used in these experiments.

Source: Reference 12

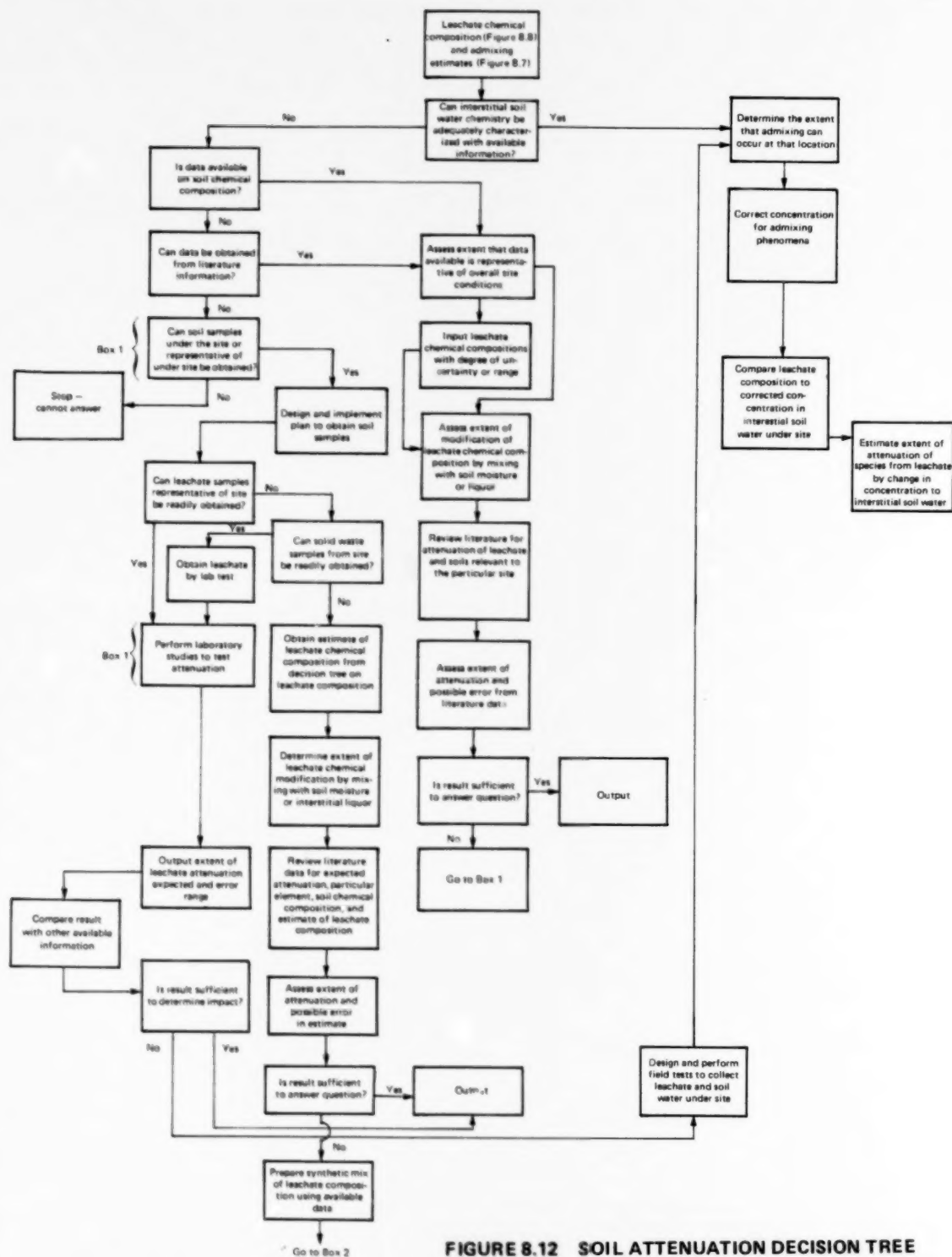


FIGURE 8.12 SOIL ATTENUATION DECISION TREE

organic carbon, Mn, Fe, Al and other major components) may be compared to those reported in the literature for specific leachates and soils to estimate the extent of attenuation.

If this procedure does not provide a satisfactory estimate, an experimental approach is then followed. Site-specific soil and leachate samples are mixed to determine changes in concentrations. If no leachate is available, leachate solutions can be prepared either by extracting waste samples or by synthetically preparing a "leachate" based on waste solid composition and literature data on leachates for similar wastes. If this laboratory approach is still insufficient, a field program that provides data on liquids under the site (the starting point in the decision-making process) should be considered.

## SECTION 9.0

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